1-1-2009

The Influence of Artificial Reef Associated Fish Assemblages and Varying Substrates On Coral Recruitment

T. Patrick Quinn

The Influence of Artificial Reef Associated Fish Assemblages and Varying Substrates On Coral Recruitment

By

T. Patrick Quinn

Dissertation submitted in partial fulfillment
of the requirements for the degree of

Doctor of Philosophy

in

Marine Biology

Nova Southeastern University

2009

COMMITTEE APPROVAL:

Major Professor

____

Dr. Richard E. Spieler
Nova Southeastern University
Oceanographic Center

Committee Members

____

Dr. Richard E. Dodge
Nova Southeastern University
Oceanographic Center

____

Dr. David S. Gilliam
Nova Southeastern University
Oceanographic Center

____

Dr. Robin L. Sherman
Nova Southeastern University
Farquhar College of Arts and Sciences

____

Dr. Kenneth E. Banks
Broward County
Environmental Protection and Growth Management Department

Abstract

This study examined enhancing coral recruitment to artificial substrate by manipulating fish assemblages and the use of coral attractant substrates. One hundred sixty artificial reef modules were organized into 40 four-module replicate configurations (quads) of varying complexity to induce different fish assemblages. The deployment array consisted of the 40 quads, each in a square configuration with three to four-meter sides (approximately 1 m separation between modules) measured from the outside corners. The quads were divided into four fill treatments of differing complexity: Empty, Small, Mixed, and Large. Each quad had four potential coral attractant treatments on settlement plates: $CaCO_3$, iron, coral transplants, and control. Each module in a quad contained a different attractant. Fish counts were conducted quarterly (January, April, July, October) for three years. During the study, fishes comprised of 166 species from 40 families were counted. Twenty-six species accounted for 90% of the fish counted with bluehead wrasse (*Thalassoma bifasciatum*), juvenile grunts (*Haemulon* spp.), and slippery dicks (*Halichoeres bivitattus*) making up over 55% of the fishes counted. Fish abundance and species richness were significantly less on Empty treatment quads than the other three treatments while species richness was less on the Empty and Small treatments than the Mixed and Large. Because of low coral recruitment rates, a single survey was conducted at the end of the study period to record the number and species of coral recruits. A total of 186 coral recruits were counted on a sub-sample of modules. *Porites astreoides* was the most abundant recruit (47.8%) followed by *Agaricia agaricites* (13.4%). Coral recruits were categorized by size and, based on an assumed 12 mm/yr^{-1} coral growth rate, separated into year classes post reef deployment. Size classes were then compared with fish abundance data. Correlations were found with Year 1 coral

recruits and damselfishes (Pomacentridae), reef butterflyfish (*Chaetodon sedentarius*), and grunts (*Haemulon* spp.). Additionally, correlations were found between Year 3 recruits and all fish species combined, and between Year 4 recruits and reef butterflyfish. Thirty coral recruits were counted on the settlement plates, with *P. astreoides* making up over 63% of the recruits. Due to the low number, rigorous statistical analysis could not be performed on the data; however, $CaCO_3$ plates had almost twice the number of recruits than the other attractants. Recommendations from this study include design of artificial reef with holes and shadowed refuge, placement of reef near natural hard-bottom or reef, and use of limestone aggregate to enhance coral recruitment. Additionally, coral transplantation may be an effective coral recruit attractant, but care should be taken in transplant species selection and collection methodology.

Table of Contents

List of Figures

List of Tables

Acknowledgements

I thank my advisor Dr. Richard Spieler and my committee members Drs. Richard Dodge, David Gilliam, Robin Sherman, and Kenneth Banks. They have all become good friends over the years and we have shared many adventures, both near and far.

I thank the many students that helped in this project from initial construction of the reef balls to the final data collection: Courtney Arena, Dr. Paul Arena, Jeremy Barnes, Jennifer Becker, Rob Baron, Neil Baertlein, Bethany Basten, Tonya Bonilla, David Bryan, Brian Buskirk, Jen Campa, Meaghan Darcy, Brian Ettinger, Danny Fahy, Kristi Foster, Jessica Freeman, Shawn Gill, Amy Hall, Heather Halter, Fleur Hartung nee Ferro, Dr. Kevin Helmle, Arlo Hemphill, Michael Hoke, Lance Jordan, Jamie Monty Vernacchio, Dr. Ryan Moyer, Frasier Meckle, Ashley Noyd, Stefanie Ouelette, Bob Patterson, Abby Renegar, Catrina Robinson, Judy Robinson, Lance Robinson, Lauren Shuman, Susan Thornton, Catherine Vogel, Dr. Brian Walker. I apologize if I have inadvertently omitted anyone.

Thanks to the many students who have been part of the Fish Lab during the course of this study. Their continuous help over the years building the modules and settlement plates, adding fill, attaching plates, and doing the assessments really made it a team effort.

Special thanks to Elizabeth Glynn Fahy who was ‘dumped on’ by me at the very beginning of the construction process and went from Master’s student to project manager overnight. We spent more than a few hours underwater together which ran the spectrum from wordless arguments to choking from a flooded mask numerous times by her ability to make me laugh.

To Dr. Gilliam and the NCRI team that helped do the coral assessment. Without their expertise and time commitment, the task of assessing the quads for coral recruits would have been a virtually impossible task.

Thanks to Kevin Kohler for writing the program ReefPlot. It became a vital tool and was used from the very beginning to locate all the quads and through hundreds of dives to the last fish and coral counts.

Many thanks go to Kathy Maxson for always coming though in a miraculous way to obtain literature and reprints that seemed impossible to find. Without her help, patience, and expertise, this dissertation would not have been completed.

Broward County personnel: Dr. Kenneth Banks, Pamela Fletcher, Louis Fisher, Joseph Ligas, and David Stout provided extensive assistance in the planning, deployment, and repositioning of quads.

Finally, many thanks and appreciation to my wife Angela and my two children Mary and Evan who have tolerated my quest for this goal for many years.

This research was funded in part by a contract with Broward County Environmental Planning and Growth Management Department and by the National Oceanic and Atmospheric Administration Coastal Ocean Program under awards NA160A1443 to Nova Southeastern University for the National Coral Reef Institute.

1. Introduction

Coral reefs are in a state of decline around the world due to natural (global warming, tropical storm systems, tsunamis, disease, and predation) and anthropogenic (coral mining, sedimentation, blast fishing, nutrient loading, over-fishing, ship groundings, divers/anchors) causes (Edwards and Gomez 2007). Recovery times for a damaged reef may be years to decades or longer depending on type, intensity, duration of the stress, and the life histories of resident species (Kinsey 1988; Johnson and Preece 1992; Riegl and Luke 1998; Jaap 2000). While the use of generalized management practices over large areas will be required to address many threats to coral reefs, physical damage from some of the more isolated causes (e.g. ship groundings, anchor placement, and blast fishing) may be appropriately dealt with in a localized setting using restoration techniques. Due to the severe destructive nature of some of these impacts, restoration is often not only an option, but is required to stabilize the reef structure, thus preventing further damage. Restoration may also be required to replace habitat structure in attempts to bring the system back to a pre-disturbed condition (Clark and Edwards 1994; Frid and Clark 1999). Much coral reef restoration work has been focused directly on restoring the physical structure of damaged reefs (Grove et al. 1980; Clark and Edwards 1994), coral transplantation or reattachment (Hudson and Diaz 1988; Jaap 2000), and not on research. Thus, little is understood, relative to restoration, concerning the dynamic interaction of colonizing biota and the substrate on which they are found (Spieler et al. 2001).

Restoration can be divided into various categories that range between extremes. At one end of the range is a costly and labor intensive attempt to return the disturbed ecosystem to a pre-damaged condition which would be self-maintaining and have a predisturbance ecological value. At the other end of the range is not performing any restoration activity and allowing the ecosystem recovery to be left to natural processes (Cairns 1991). The former may be impossible to achieve without the predictive capability to know the ecological outcome of restoration efforts, and the latter may result in an undesirable phase shift (i.e. coral and gorgonian dominated system to an algal turf dominated system) (Mumby et al. 2007). Most restoration projects fall in between these two extremes.

Coral reef restoration is an expensive process and costs can range from \$100,000 - \$1,000,000^{+} U.S. per hectare while the estimated average annual value of coral reef goods and services is \$6,075 U.S. per hectare (Edwards and Gomez 2007). Thus, it is vitally important from an economical, as well as an ecological perspective, to ensure that restoration projects are efficient, effective, and designed specifically for the habitat which is to be restored.

It has been recommended that specific goals be established for determining the reef's return to health (Sheppard et al. 2000). Normally, the major goal would be restoration of the ecological functions, in terms of species richness and abundance, in as short a time period as possible. In order to reduce recovery time of an affected reef, active restoration (influencing the course of recovery by direct intervention) might be used to increase the abundance of desirable species (Woodley and Clark 1989). However, little is known about inter-specific interactions that may influence the success of such restoration efforts.

Some authors use the term rehabilitation for accelerating recovery to an endpoint and reserve the term restoration for a full return to pre-impacted conditions (Pratt 1994; Pickering et al. 1998; Zarull and Hartig 2001; Precht and Robbart 2006); however, others do not make this distinction (Hackney 2000; Jaap 2000; Yap 2000; Weinstein et al. 2001) and it will not be made here.

1.1 Background

1.1.1 Artificial reefs and restoration

Artificial reefs are often used during primary restoration efforts to return structure to damaged reefs. Artificial reefs can offer multiple returns on investment by simultaneously stabilizing loose substrate, providing refuge for fish, and functioning as structure for benthic community development or for coral transplantation (Clark and Edwards 1994). Examples of artificial reefs used to rebuild damaged reef framework include concrete structures and/or limestone boulders while flexible concrete mat is often used for stabilization of loose material (Clark and Edwards 1999; Jaap 2000). Another example of the use of artificial reefs for restoration is the deployment of structures as a deterrent of extractive practices (e.g. trawling) to allow a habitat to recover from chronic destructive practices (Pickering et al. 1998).

Another use of artificial reefs is as a tool to examine the ecological processes occurring in the marine environment for application during a restoration effort. The focus of these studies is usually singular, examining either fish assemblages (Ambrose and Swarbrick 1989; Kellison and Sedberry 1998; Golani and Diamant 1999; Rilov and Benayahu 2002; Brickhill et al. 2005) or benthic (coral/algal) communities (Baynes and Szmant 1989; Fitzhardinge and Bailey-Brock 1989; Relini et al. 1994; Perkol-Finkel et al. 2008), but recent studies have begun to examine multiple processes on artificial reefs involving both vertebrate and invertebrate species (Cummings 1994; Abelson and Shlesinger 2002; Thanner et al. 2006).

Additionally, artificial reefs have been used to focus on a narrow aspect of the ecological processes such as fish and mobile invertebrate recruitment (Butler IV et al. 1995;

Herrnkind et al. 1997; Gilliam 1999; Sherman et al. 1999; Sherman 2000; Sherman et al. 2001), coral and benthic assemblage recruitment (Baynes and Szmant 1989; Perkol-Finkel and Benayahu 2007), predation and refuge (Gilliam 1999; Sherman et al. 2001; Hixon and Jones 2005), and foraging behavior (Bortone 1999). Artificial reefs are useful for these types of studies. With specific design many of the variables (i.e. rugosity, complexity, and micro-habitat) found on natural reefs and even small patch reefs can be controlled. A problem however is the range of materials and structures used in research. The artificial reef types used vary from piles of conch shells (Shulman 1985a, b; Beets 1989), limestone or quarry rock (Ambrose and Swarbrick 1989; Cummings 1994; Abelson and Shlesinger 2002), concrete block in a uniform arrangement (Bohnsack and Talbot 1980; Bohnsack 1983; Hixon and Beets 1989, 1993; Carr and Hixon 1997), or block piles (Ogden and Ebersole 1981; Brock and Kam 1994) up to large manufactured structures (Clark and Edwards 1994; Frazer and Lindberg 1994; Jara and Cespedes 1994; Eklund 1996; Clark and Edwards 1999; Gilliam 1999; Sherman et al. 2001, 2002; Thanner et al. 2006), ships (Chandler et al. 1985; Arena 2005), tires (Haughton and Aiken 1989), and other materials that include metal, plastic, and PVC (Alevizon and Gorham 1989; Bortone et al. 1994; Gregg 1995; Golani and Diamant 1999). This wide range in artificial habitat design composition among the artificial reef studies makes the transfer of knowledge from academic study to practical use a difficult prospect.

1.1.2 Fish/Coral interactions

Reef fishes can influence settlement and growth of coral species in various direct and indirect ways. These influences may include incidental consumption of newly settled corals or small coral colonies by large herbivores (acanthurids and scarids) (Randall 1974) and direct predation by fish corallivores (pomacanthids and chaetodontids) (Hourigan 1988). Hourigan (1988) observed feeding rates up to 710 bites/hr by two species of butterflyfish in the Caribbean while Gochfeld (1991) counted feeding rates up to 240 bites/hr by the coral-feeding damselfish *Plectroglyphidodon johnstonianus* off Oahu, Hawai'i. Additionally, territorial damselfish, establishing algal gardens, can exclude grazers which has the effect of allowing greater coral recruitment (Sammarco and Carleton 1981) or providing incidental protection of larger hermatypic corals on the periphery of territories (Wellington 1982). However, territoriality among fish causing the exclusion of large herbivores has also been shown to result in death of smaller coral colonies due to competition from an increased standing crop of algae (Sutton 1983).

Interestingly, stoplight parrotfish (*Sparisoma viride*) have been documented off Southeast Florida feeding on transplanted *Siderastrea siderea* colonies (Brownlee et al. 2008), but it is not clear if this occurs on non-transplanted corals. In the Florida Keys parrotfishes (*Sparisoma* spp.) were observed feeding directly on transplanted *Porites divaracata* and *P. porites* colonies (Miller and Hay 1998) while queen triggerfish (*Baliste vetula*) have been observed biting off pieces of *Agaricia tenuifolia* in the Bahamas (K. Banks, personal communication).

Further, fishes can affect coral growth by increasing local nutrient levels. Fish excretory and fecal products comprise a substantial source of nitrogen and phosphorous on coral reefs (Meyer et al. 1983; Meyer and Schultz 1985a). Concentrations of NH_4^+ were more than four times higher around coral heads with resting schools of fish and coral growth rates were significantly higher on coral heads with grunts (Meyer et al. 1983).

1.1.3 Coral recruitment

The rate of coral growth on a reef is initially dependent on the quantity of larvae that settle on the substrate (Johnson and Preece 1992). While preferential settlement has been shown to occur on various surfaces, e.g. concrete, metal, quarry rock, tires, and red coralline algae (Fitzhardinge and Bailey-Brock 1989; Morse and Morse 1996), it is often unclear as to what type of stimuli (physical, biological, chemical) are influencing the larval settlement. For example, it has been determined that some scleractinian corals possess substances (allelochemicals/larvotoxins) that can adversely affect the settlement of competitive hermatypic species (Fearon and Cameron 1997).

Due to the high mortality of coral larvae from the time of settlement to observation (Fitzhardinge and Bailey-Brock 1989) and the ability of the larvae to reject the substrate and detach itself (Mullineaux and Butman 1991; Reyes and Yap 2001) the measure of success of coral recruitment in reef restoration should focus, not on the number of initial settlers, but on recruit survivorship, i.e. juvenile corals that have reached a specific size (Rodriquez et al. 1993; Atrigenio and Alino 1995). Goreau et al. (1981) estimated non-density dependent mortality of *Porites porites* planula at over 90% with post-recruit death by browsers scraping hard substrate for algae and interspecific spatial competition likely to be major causes of mortality of juveniles; however, Birkeland (1997) determined that Caribbean fish will avoid feeding on corals as small as 2.5 mm in diameter.

1.2 Scope and Purpose

Coral reef restoration plans can include structural enhancement with artificial reefs for habitat loss due to physical damage (e.g. ship groundings). There have been more than 11 ship groundings on the reefs off Broward County in the past 15 years (Banks et al. 2008).

The design criteria for restoring such impacted reefs does not exist due to lack of scientific data. Grounding sites from large vessels can cover relatively large areas of coral reef habitat; however, most studies use single artificial reef modules to examine ecological processes that may be happening on the much larger nearby reefs.

Doherty and William (1988) stated it is doubtful that small experimental unit results can be extrapolated meaningfully to large complex habitats. Previous studies off Broward County have used small (1 m^3) modules to study fish recruitment processes (Gilliam 1999; Sherman et al. 1999, 2001, 2002) while only one study has used multiple reefs (Jordan et al. 2005). Deis and Kosmynin (in press) used large (> 1 m^3) artificial reef structures to examine coral recruitment, but fish assemblages were not examined. Eklund (1996) used larger structures, 2.4 m^2 (base) x 1.8 m high to examine fish predation and resource limitation. However, her study was conducted in Palm Beach County, located north of Broward County, with less reef structure (Banks et al. 2007) and a different fish community than Broward County (DERM). My study used multiple (4) artificial reef modules placed close together (1 m separation) on the assumption that the modules would function as one artificial reef with a large "footprint" (13 m^2).

The goal of this research was to examine the potential use of artificial reef structures to provide reconstructive structural complexity and refuge to fish assemblages and in turn enhance the number of corals recruiting to the structures. Small identical concrete artificial reefs were used as replicates to eliminate any confounding effects that may be found using other more variable material commonly deployed as artificial reefs. Replicates were used and deployed in a similar environment such that any differences between the reefs would be attributed to design manipulation. Through manipulation of the modules' internal complexity, different fish assemblages were expected to develop. Habitat structural complexity has been shown to contribute to an increase in the diversity of fish on natural as well as artificial reefs (Roberts and Ormond 1987; Caselle and Warner 1996; Sherman et al. 2001, 2002; Shima et al. 2008). A relationship between fish and corals on reefs has been demonstrated both positive (Meyer et al. 1983; Bell and Galzin 1984; Meyer and Schultz 1985b; Mumby et al. 2007) and negative (Randall 1974; Neudecker 1979). Whether the relationship is merely a result of corals creating more complexity remains unclear (Chabanet et al. 1997) although Holbrook et al. (2008) found a relationship between fish (species richness, total abundance, and species composition) and live coral.

1.3 Hypotheses

Three specific hypotheses were developed to answer questions regarding fish assemblages and coral recruitment.

1.3.1 Hypothesis 1

1) Complexity (4 treatments)

H_1: Fish assemblages associated with artificial reefs result from a difference in the artificial reef structural complexity.

Inference$_1$: If H_1 is correct, then artificial reefs constructed with differing refuge (hole) sizes should acquire different fish assemblages.

1.3.2 Hypothesis 2

2) Fish community/coral recruitment interaction

H_2: Different fish assemblages affect the recruitment of coral onto artificial reefs.

Inference$_2$: If H_2 is correct, then artificial reefs with different fish assemblages should have varying coral recruitment in terms of abundance and species.

1.3.3 Hypothesis 3

3) Attractants (4 treatments)

H_3: Coral recruitment to settlement plates can be influenced by substrates or attractants.

Inference$_3$: If H_3 is correct, then settlement plates treated with substrates/attractants should have greater coral recruitment than non-treated settlement plates.

Comparing fish assemblages among artificial reef modules containing different degrees of complexity will test Inference$_1$ that artificial reefs constructed with differing refuge (hole) sizes should acquire different fish assemblages. If differences exist among the complexity treatments, my first hypothesis (H_1), that fish assemblages associated with artificial reefs are affected by the structural complexity, would be supported. If not, I would reject H_1. In either case the results would yield important information for the design of artificial reefs intended for use by fish.

Comparing fish assemblages with coral recruitment will test Inference$_3$ that different fish assemblages play a role in the structure of a coral community. If coral recruitment differences exist on artificial reefs with different assemblages, my second hypothesis (H_2) would be supported in that coral recruitment is affected by the structure of the fish assemblages associated with it. If there are no differences, then I would reject H_2. The results should ideally yield important information concerning the design of artificial reefs intended to develop specific fish and coral communities.

Comparing coral recruitment to settlement plates treated with substrates/attractants will test Inference$_2$ that applied substrates can be used to induce greater coral recruitment. If recruitment differences exist between treatments, my third hypothesis (H_3), that coral recruitment can be affected by the use of attractants, would be supported. If not, I would reject H_3. The results would yield important information for the use of the tested attractants with artificial reefs intended as substrate for a coral community.

2. Materials and Methodologies

2.1 Study Site

The marine environment off Broward County consists of an inner reef hard bottom ridge area and three (inner, middle, outer) reef terraces (Moyer et al. 2003). These terraces have been described as a drowned Holocene reef (Lighty et al. 1978) or Pleistocene bedrock covered by a coral veneer (Goldberg 1973).

The study site chosen for the deployment of the artificial reef array was on a sand area in 13 m of water between the middle and outer reef terraces off Dania Beach, FL (Figure 1). These adjacent terraces have a coral-reef-associated ecosystem dominated by octocorals (Goldberg 1973; Banks et al. 2008). The artificial reef array was planned as three parallel lines running North-South between the reef tracts, 30 m from any natural reef, hard bottom, or other artificial reef (Figure 1). Other artificial reefs in the area are located between the array and the middle (western) reef tract (Figure 1). These include the ATT/DERM modules, Warren modules (not shown in Figure 1), and a limestone boulders/concrete tetrahedron pile locally known as Mt. Dania.

2.2 Experimental Design

One hundred sixty artificial reef modules were organized into 40 four-module replicate configurations (quads). Within each quad, each module had varying complexity to induce different fish assemblages. Each of the four modules within each quad had one of four fill treatments: empty, small, mixed, and large. The deployment array consisted of the 40 quads each in a square configuration with three to four-meter sides (approximately 1 m

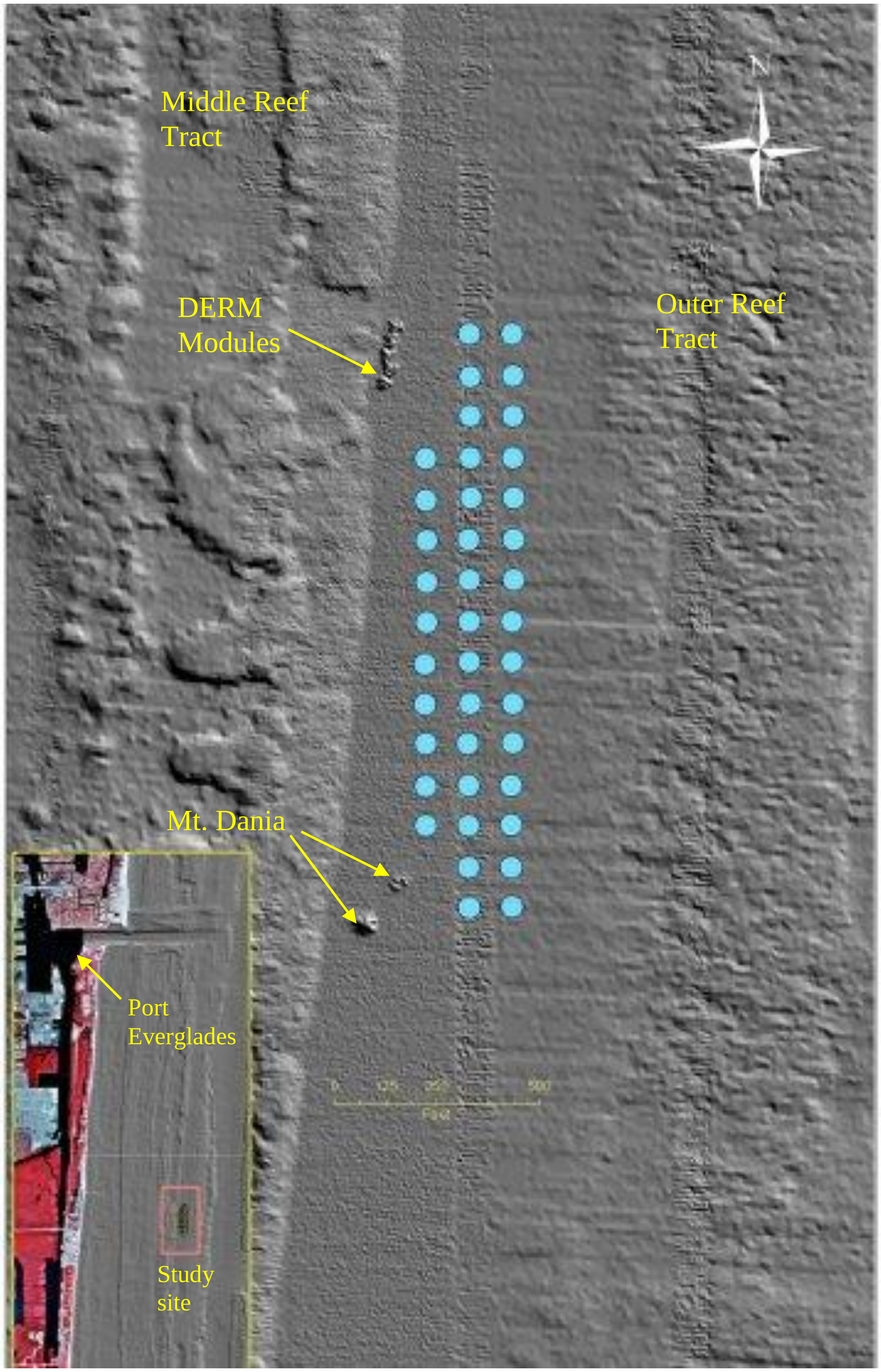

Figure 1: Study site and proposed quad array. Each circle represents one quad (four modules). Inset shows study site relative to Port Everglades.

separation between modules) measured from the outside corners. A transplant module (modified to receive two coral transplant cores) was one of the four modules in each quad. One module of each quad was marked with numbered tags to allow for underwater identification. For within-quad tracking purposes, modules in a quad, as viewed from above, starting with the transplant module and moving in a clock-wise direction, were designated M1 (transplant), M2, M3, and M4.

Additionally, one of each module had one of four coral recruit settlement treatments. Settlement plates were treated with three different types of potential coral attractants: $CaCO_3$, iron, and coral transplants (as used here, attractant may be either a chemical, biological, or physical stimulus for coral settlement) and attached to the modules. $CaCO_3$ (limestone) is a common component used in reef restoration and studies have found $CaCO_3$ may be a favorable substrate for coral recruitment (Wallace and Bull 1981; Harriott and Fisk 1987; Hudson and Diaz 1988; Scott et al. 1988; Miller and Barimo 2001). Vessels or other metal structures are often deployed as artificial reefs and metal may be a suitable component for coral recruitment (Fitzharding and Bailey-Brock 1989) and artificial vessel reef structures found in the coastal waters off Broward County have a diverse and abundant coral community (author, personal observation). Finally, some species of coral have been shown to settle near larger colonies of the same species (Lewis 1974) and coral transplantation has been suggested as a way to stimulate coral growth onto reefs (Oren and Benyahu 1997). Thus the experimental variables to be examined here are: complexity, use of potential coral attractants, and fish assemblage/coral recruitment interaction.

2.3 Artificial Reef Design, Construction, and Deployment

During July and August 2000, 160 small artificial reef (Pallet Balls™) modules were constructed and staged at the Nova Southeastern University Oceanographic Center (NSUOC). The modules were 1.3 m wide at the base, one meter tall, open at the top, and had a large central void space. Additionally each module had 15 side holes which were arranged in a consistent pattern for all modules.

Fifteen molds were used daily during the construction process and followed established protocols as developed by the Reef Ball Development Group, Ltd. Molds were assembled and prepped during the morning, and the concrete (aggregate size 0.6-1.3 cm) poured during the early afternoon hours and allowed to harden overnight. Molds were removed the following morning and the exterior surface of the modules was washed with a high pressure water hose to expose the aggregate. This created a rough external surface thought to facilitate coral recruitment and survival. Representatives from the Reef Ball Development Group oversaw construction with manual labor supplied primarily by student volunteers.

Forty modules were modified slightly during construction to accept two 10 cm diameter coral transplants. The modification consisted of two 10 cm diameter holes cut into the module molds and plastic cups inserted prior to the concrete being poured (Figure 2). Once removed, the cup inserts left a 10-cm cylindrical depression designed to attach coral transplants. These 40 modules are referred to here as transplant modules. On November 16, 2000 the modules were loaded onto a barge using a crane at the Navy Surface Warfare Center in Dania Beach FL, adjacent to NSUOC at Port Everglades with deployment on November 17, 2000. Individual quad sites were located with Differential Global Positioning

System (DGPS) and marked using buoys deployed by the Broward County (Florida) Environmental Protection and Growth Management Department (EPGMD). Once buoys were in place, the barge attempted to position next to a buoy allowing a crane to deploy one quad (four modules) at a time onto the site (Figure 3). However, due to weather conditions, the barge was unable to stay on the specific buoy locations, and this resulted in the majority of quads being deployed off planned sites.

Between November 2000 and January 2001, efforts were made to locate the quads and obtain differential global positioning system (DGPS) coordinates, however, after 12 dives, only 38 quads were located. On January 6, 2001, calm seas and clear water conditions allowed for a two hour systematic survey by boat of the array area. Repeated passes were made over the array and DGPS coordinates were taken each time the boat passed over a quad. Coordinates were then entered into ReefPlot, a mapping program developed by Kevin Kohler at NSUOC, which created a grid map showing each quad's location. On January 11-12, 2001, divers used laminated copies of the ReefPlot map as navigation aids to resurvey the area and tag quads. This ensured all 40 quads had been located, and confirmed all modules were accounted for and undamaged. Additionally, more specific DGPS coordinates were obtained by divers securing a dive flag down-line in the middle of each quad which placed the actual flag directly above the modules. The boat then came alongside the flag to record the position. When the more accurate DGPS coordinates were entered into ReefPlot, it became apparent that the majority of the quads were out of position and would have to be moved.

Figure 2: Module mold with two 10 cm diameter plastic cup inserts used to create recesses in the module where coral transplants were later inserted. Cup inserts were removed with the mold.

Figure 3: Module deployment by crane from a barge in groups of four (a quad) onto specific sites marked by buoys.

In conjunction with EPGMD, an assembly of surface lift bags, cables, pulley/crank, and straps was devised to relocate incorrectly positioned quads. The assembly consisted of lifting-strap ends threaded through modules' side holes, from the inside out, and held in place with a steel bar. The middle of the lifting-straps came up through the modules' top opening and connected to the lower end of pulley/crank. The upper end of the pulley/crank was connected to cables that extended to the surface and then joined with three 2-ton pillow lift bags. Once all four modules in a quad were connected this way, each separated by a wooden 2x4 spreader-bar attached to the cables, the pulley/cranks were used to raise each module approximately 30-40 cm off the bottom (Figure 4). A tow-line connected to the first module extended approximately 10 m past the targeted end location, through a snatch block connected to a trailer-screw driven in the sand, and then up to the surface for retrieval by the boat. When all the modules were lifted, communicated to the boat by divers using buoy signals, the assembly was towed to the target location and the modules were repositioned in a square configuration.

The relocation effort began in March 2001 and was completed on June 2001. The originally planned array of three parallel lines was unable to be achieved as it would have required almost all of the quads to be moved. Instead, a compromise array was created to try and maximize distance between quads and other natural or man-made structures while minimizing the number of quads requiring movement. Eventually, 20 quads (80 reef ball modules) were moved to achieve the accepted array (Figure 5). Final spacing between quads ranged from 15-35 m.

Arrangement of the quad to the final square configuration required a 1-meter separation distance between reef ball modules. Any quad modules out of alignment within its square configuration were moved by attaching multiple small lift bags to the modules (Figure 6). This allowed divers to physically slide the modules into the proper spacing configuration. Alignment of modules within quads was done from February through June 2001.

Figure 4: Quad relocating with lifting straps and lift bags. Divers connected lifting-straps through the modules' side holes, up through the center opening and connected the straps' middle to pulley/cranks. Each pulley/crank was then connected to a cable extending to the surface where it joined with a pillow lift bag.

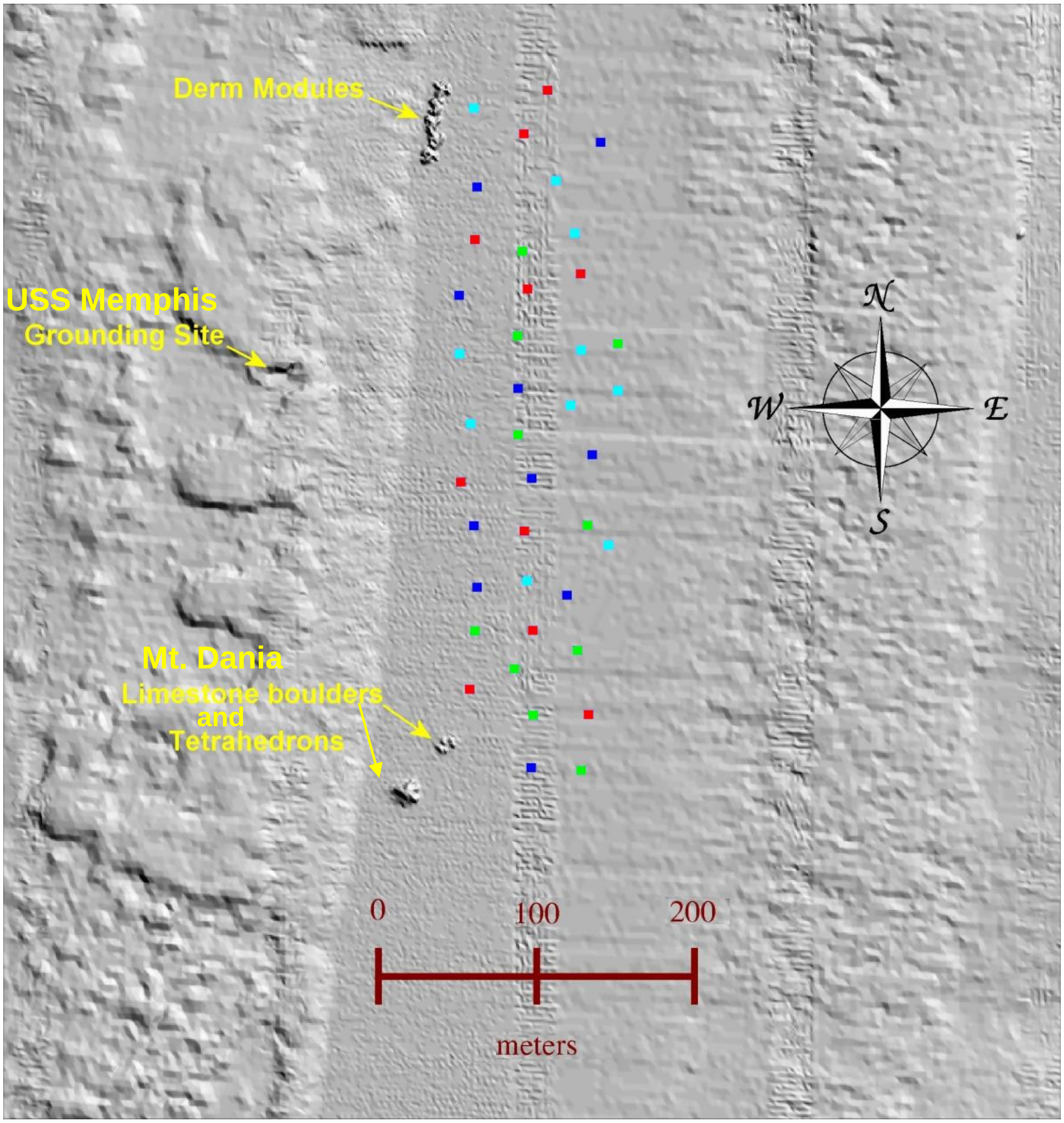

Figure 5: Final array of quads after relocation effort. Each colored square is the location of an individual quad and different colors designate different quad internal complexity: green = empty, blue = small, aqua = mixed, red = large. Yellow annotation shows locations of other artificial reefs in the area along with the grounding site of the USS Memphis.

Figure 6: Divers repositioned individual modules within a quad. This was done to ensure a minimum 1-meter separation between modules required for a square quad configuration.

2.3.1 Internal complexity

The 40 quads were randomly divided into four treatment types of internal structural complexity: empty, small fill, mixed fill, and large fill. This resulted in 10 replicates of each treatment. The 10 Empty treatment quads did not have any fill material added to the central void space of the modules.

Small fill quads were created by adding a plastic mesh cone (2 cm^2 mesh grid size) into each module's central void space (Figure 7). The mesh cone was open at the bottom, large enough to cover the inside base of the module and extend to the top, and was secured using large cable ties through the bottom holes of the module. Forty mesh cones were used to create internal complexity in the Small fill treatment.

The third set was a Mixed fill treatment. One module in a Mixed treatment quad was left empty, two modules received mesh cones, and the final module received 4 concrete block (Figure 8). A total of 20 mesh cones and 40 concrete block were used in the ten Mixed quads.

The remaining ten quads were used for the Large treatment having the void space in each module filled with four concrete block. One hundred and sixty concrete blocks were used (four blocks per module,16 blocks total for a quad) to create the large fill treatment quads. Blocks were transported via boat to a quad's DGPS coordinates where the appropriate number of block for that quad were dropped overboard. Divers later collected the blocks and placed them inside the appropriate modules, positioned so the holes were open to the vertical side and thus accessible to fish and invertebrates. The addition of cage material and concrete blocks took place from May to July 2001.

Figure 7: Plastic mesh cone (2 cm^2 grid) placed in central void space to function as small fill. Each cone was secured using plastic tie-wraps through the module's bottom holes.

Figure 8: Concrete blocks placed in central void space to function as large fill. Each block was placed with the holes open on the vertical side. Four blocks were placed in each large fill module.

2.3.2 Settlement plates – Coral attractants

Three hundred and twenty 30 cm^2 settlement plates for use on the modules were constructed at the same time and with the same concrete mixture as the modules. Plates were formed using two wooden grids, each outlining twenty-one 30x30x2 cm areas, assembled as frameworks on top of plywood sheet bases (Figure 9). The wooden grid frameworks were assembled each morning and, prior to the addition of concrete, the frameworks and plywood bases were sprayed with vegetable oil to help prevent plates from adhering to the wood. After the module molds were filled each day, the remaining concrete was poured into wheel barrows and shoveled by hand into the grid outlines. Plates were allowed to harden overnight, removed each morning for on-site curing and storage, then frameworks prepared for the next pour.

After curing, all settlement plates were individually coated with a thin layer of concrete to allow the coral attractant treatments to be applied. The concrete coating was mixed with micro-silica to closely mimic the concrete mixture used for construction of the plates and modules. During this coating process, 80 plates had a layer of $CaCO_3$ (crushed limestone) spread over the surface while the thin concrete layer was still wet (Figure 10). A four-inch diameter length of PVC pipe was used as a rolling pin to embed the limestone into the wet concrete. The large pipe diameter helped ensure pressure required to embed the limestone would not adversely affect the thickness of the concrete layer. Due to the physical nature (variable grain size) of the crushed limestone, it was impossible to quantify the amount of CaCO3 successfully applied to each settlement plate. After the concrete layer was dry, the settlement plates were gently brushed to remove any loose crushed limestone.

Figure 9: Construction of settlement plates. Plates were completed at the same time and with the same concrete mixture as the artificial reef modules.

Another 80 plates were treated in a similar fashion with 20 g of 1-2 mm diameter iron granules (Alfa Aesar, stock # 39708) (Figure 10). The remaining 160 plates were also coated with the thin concrete layer, but otherwise left untreated: 80 for transplant modules and 80 to serve as controls.

Although each module was eventually fitted with two settlement plates, to prevent plate damage, the plates were not attached until after quad deployment. Settlement plates were stored at NSUOC until deployment on July 30-31, 2001. Plates were transported to the array via boat and lowered, eight plates at a time (two with $CaCO_3$ treatment, two with iron treatment, and four with plain concrete coating), to divers waiting at a quad. Plates, in pairs of identical treatments, were temporarily positioned upright on the sand substrate leaning against the outward facing side of each module. Treatments were placed randomly within the quad, the exception being to ensure untreated plates were placed on the transplant module.

All 320 settlement plates were placed in the water with quads prior to any plates being attached to the modules. Divers first used a wire mesh brush to clean the top and side module surface of any sediment or colonizing organisms where the settlement plates were to be attached (Figure 11) then applied a pre-mixed cement blend to attach the settlement plate. The blend was an equal mixture of Portland Type II cement and white molding plaster, which was then combined with silica sand in a 3/1 mixture/sand ratio (R. Galletta, Industrial Divers Corporation, personal communication). The addition of molding plaster decreased the amount of time the blend required to harden.

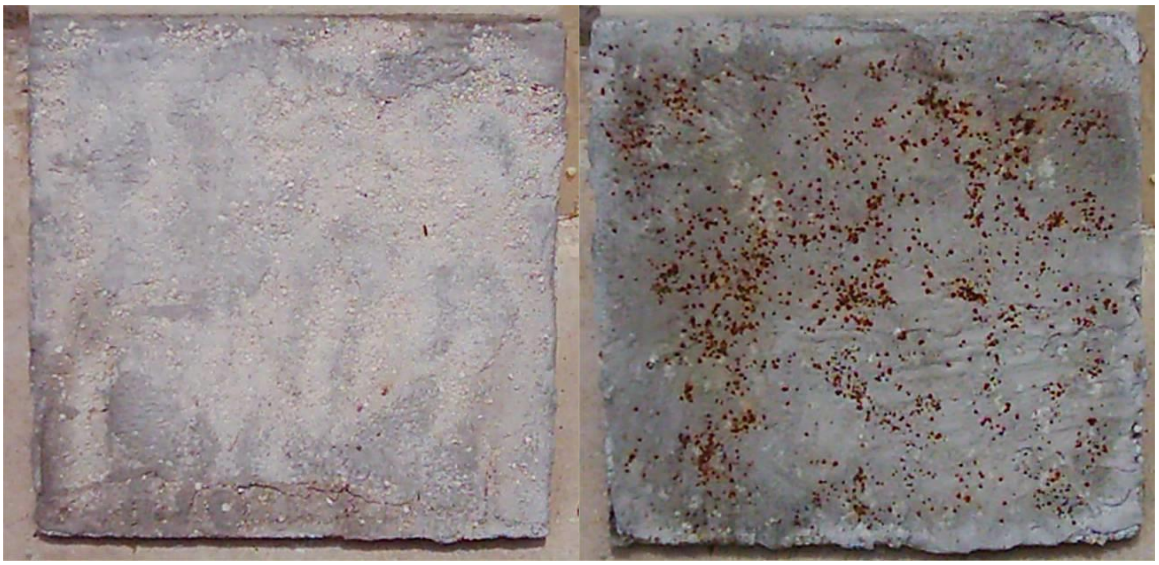

Figure 10: A thin layer of micro-silica concrete mixture was spread over each settlement plate. Eighty plates had $CaCO_3$ embedded into the layer (left), 80 plates had iron granules embedded in the layer (right), while the remaining 160 plates were only coated with the mixture (not shown) and used on the transplant modules or as controls.

Divers carried measured amounts of the blend in plastic zip lock bags. Once the module surface was cleaned, the diver opened one end of a bag to let in enough water to put the blend into solution, resealed the bag, and kneaded the blend solution until it reached the consistency of clay and felt warm to the touch. With the blend at the appropriate consistency, the diver scooped out the blend, applied it to the module, and pressed the settlement plate onto the blend mass (Figure 12).

Figure 11: Prior to attaching settlement plants, divers used a wire brush to clean the module surface of any sediment or newly settled organisms where the cement was to be applied.

Figure 12: Settlement plates were attached with a concrete/plaster blend to the top and sides of each module.

2.3.3 Coral attractants

Four different types of potential substrates/attractants were used on settlement plates to induce coral recruitment: $CaCO_3$, iron, coral transplants, and untreated concrete (controls). Prior to plate deployment, the settlement surface of 160 plates was treated with a potential coral attractant; 80 plates with iron granules and 80 with $CaCO_3$ (crushed quarry limestone). The remaining 160 plates were left untreated for attachment onto transplant modules where corals were used as attractants or for use as controls.

Montastrea cavernosa and *Meandrina meandrites* were chosen as coral transplant species based on their growth, survivorship and transplantation success (Fahy 2003), abundance in Broward County (Gilliam et al. 2004), and availability on the adjacent reef site near the deployment array. Between January and March, 2001, 20 donor colonies of each species were located, mapped, and tagged on the inner reef tract west of the array at a depth of approximately 9 m. Colonies were chosen that had a minimum living tissue diameter of 40 cm and appeared to be free of disease, bleaching, or substantial partial mortality. This allowed for two 10 cm diameter coral cores to be taken from each donor colony, thus reducing the number of donors required,. Also, the minimum colony size was chosen to help reduce the chance of subsequent donor colony mortality which can be inversely related to colony size (Hughes and Jackson 1980, 1985; Hughes and Connell 1987; Soong 1993). Finally, donor colonies of the minimum size would ensure both donor and transplant corals would be of reproductive size and age (Szmant 1991).

A Stanley® hydraulic drill and power pack unit with a 10-cm diameter core barrel was used to remove two replicate coral cores from each donor colony. Concrete plugs with a 10 cm diameter were then used to fill the holes left by core removal. Efforts were made to ensure the plug was flush with the surface of the donor colony then secured around the edge by a thin ‘string’ of underwater marine epoxy (Aqua-Mend®). The epoxy filled gaps between the concrete plug and coral, then was smoothed at the edges to allow for potential tissue growth over the epoxy surface and onto the concrete plug.

Once removed from the donor colony, transplant cores were placed in numbered plastic bags, transported to the surface, placed in a cooler lined with freezer packs, and cushioned with bubble wrap. The amount of time cores were in the cooler was kept to a minimum (generally less than 1 hour) before divers placed them in the cup holes on the transplant modules. Efforts were made to fit the transplant cores flush with the surface of the transplant module and, once placed in the modules, the cores were secured by using epoxy in a similar manner as the concrete plugs in the donor colonies. See Fahy (2003) for additional details regarding the coring and transplantation process. Collection of transplant corals took place from March through June 2001.

2.4 Monitoring

2.4.1 Fish assemblages

Sampling began in October 2001 and was conducted at 3-month intervals (January, April, July, and October) through 2004 with the exception of May 2004, which was substituted for April 2004 which was cancelled due to inclement weather.

The assemblage of fishes surrounding each quad were recorded using methods previously established at NSUOC for use on artificial reefs (Gilliam 1999; Sherman 2000). Divers using SCUBA recorded, on plastic slates, fish species, numbers of fish per species, and estimating total length by size class (0-2, 2-5, 5-10, 10-20, 20-30, and greater that 30 cm (30^+ cm) TL), of all fishes within 1 m of each quad.

2.4.2 Coral recruitment

A recruit is defined here as a juvenile hermatypic coral large enough to be visible with the naked eye, which for most corals is about 8-10 months after initial settlement (Harrison and Wallace 1990). Previous studies have placed visible recruits in a range of 2-10 mm in diameter (Sheppard et al. 2000; Moulding 2006) and considered corals as juveniles if ≤40 mm (Bak and Engel 1979; Chiappone and Sullivan 1996; Edmunds 2000; Edmunds et al. 2004).

Initial plans were to census, at 3-month intervals for coral recruitment, growth, and mortality on the settlement plates. However, apparent recruitment was so low that only a final census was conducted in 2004 on all settlement plates and a sub-sample of quad modules. The sub-sample regimen was due to the amount of time involved in counting coral recruits and consisted of counting all coral recruits on 2 specific modules per quad: Module 1 (M1: transplant ball) and Module 3 (M3: directly opposite the transplant module in the quad).

Divers initially examined each settlement plate for recruits. Using a metric ruler and magnifying glass, the diver measured the long and short axis, identified the recruit to the lowest taxonomic level possible in the field, and recorded data on underwater paper. Divers then followed the same protocol for recruits located on the outside surface (excluding the bottom) of M1 and M3 and the location of the recruit on the module was noted on a sketch-outline diagram. Census of the settlement plates and M1 was conducted in May 2004 and census of M3 was conducted in August 2004.

Due to the lack of data regarding coral recruitment on an annual bases, recruit data were organized into size classes based on a conservative growth rate estimate of 12 mm yr^{-1} (Van Moorsel 1988; Edmunds et al. 2004). Based on this growth rate, size classes of 0-12, 13-25, 26-38, 39-51, 52-64, and 65-77 were created. These size classes were used in correlation analyses with fishes.

2.5 Data analysis

Analysis of fish assemblages (abundance and species richness) associated with complexity treatments was accomplished with a mixed model analysis of variance (ANOVA) technique and a post-hoc Tukey-Kramer (TK) comparison of means using SAS V9.1 software (SAS Institute Inc. Carey, NC, USA). A probability value of less than 5% ($p < 0.05$) in both ANOVA and TK was accepted as a significant difference. The abundance data were log transformed [$\log_{10}(x + 1)$] prior to analysis (McManus et al. 1981; Zar 1996) to meet the assumptions of equal variance and normality of the ANOVA. Biomass was determined using published length-weight relationships (Bohnsack and Harper 1988). Mid-point in each size class was used as length and similar congenerics for any species with a specific length-weight relationship.

Analyses of coral recruit data were accomplished with an analysis of variance (ANOVA) technique and post-hoc Newman-Keuls (NK) test for comparison of means using Statistica 6.0. A probability value of less than 5% ($p < 0.05$) in both ANOVA and NK was accepted as a significant difference. Data were log transformed [$\log_{10}(x + 1)$] (McManus et al. 1981; Zar 1996) to meet the assumptions of the ANOVA prior to analysis.

Multivariate statistical analyses for both fish and coral were performed using the Plymouth Routines In Multivariate Ecological Research statistical package (PRIMER V6) including multi-dimensional scaling (MDS) plots of Bray-Curtis similarity indices, analysis of similarity, (ANOSIM) tests, and similarity percentages (SIMPER) (Field et al. 1982).

Natural log, square root (Zar 1996; Moulding 2007), and $\log_{10}(x + 1)$ transformations were used to in an attempt to have all data meet the assumptions of parametric analysis. Although parametric assumptions could not be obtained using any of the transformation techniques, both parametric and non-parametric (Spearman-Rank) correlation analyses were conducted on standard and $\log_{10}(x + 1)$ transformed fish assemblage and coral recruit data. When significant results were found in both parametric and non-parametric tests, only non-parametric results are presented.

3. Results

3.1 Fish Assemblages

Over the (32 month) course of this study, 440 individual fish counts were conducted on the quads. A total of 27,665 fishes from 166 species and 40 families were counted (Table 1). Of the 166 species recorded, 26 species accounted for 90% (24,915) of the total number of fishes counted.

Bluehead wrasse (*Thalassoma bifasciatum*), with 7,630 fish counted, was the most abundant species on all quads combined, followed by juvenile grunts (*Haemulon* spp.) with 4,437, and slippery dicks (*Halichoeres bivittatus*) with 3,270. These two species and one taxa group accounted for over 55% (15,337) of the total abundance.

Between treatments, Empty quads had the fewest number of fish (5,315), species (107), and families (30). Of the 166 total species, 59 species in 28 families were not found on any of the Empty treatment quads, but 9 species in 9 families were unique to the treatment (Table 1).

Bluehead wrasse was the most abundant species on the Empty treatment with 1,663 fish, followed by slippery dicks (887) and juvenile grunts (311). Together, these three species contributed 54% (2,861 fishes) to the Empty treatment total fish abundance.

Small treatment quads had a total of 7,214 fishes from 118 species and 33 families. Forty-eight species from 24 families, of the 166 total species recorded, were not found on the Small treatment, however 12 species in 10 families were unique to the treatment. Bluehead wrasse were the most abundant species of the Small treatment with 1,957 fish, followed by juvenile grunts (1,641), and slippery dicks (790). Together, these three species contributed 61% (4,388) to the Small treatment total fish abundance.

Mixed treatment quads had a total of 7,277 fishes from 116 species and 31 families. There were 50 species from 31 families not found on the Mixed treatment and 11 species in 8 families unique to the treatment. Bluehead wrasse were the most abundant species on the Mixed treatment with 1,904 fish, followed by juvenile grunts (1,167), and slippery dicks (849). Together, these three species contributed 54% (3,920 fishes) to the Mixed treatment total fish abundance.

Large treatment quads had a total of 7,859 fishes from 118 species and 33 families. Forty-eight species from 24 families were not found on the Large treatment and 12 species from 12 families were unique to this treatment. Bluehead wrasse was the most abundant species on the Large treatment with 2,106 fish, followed by juvenile grunts (1,318), and slippery dicks (744). These three species contributed 53% (4,168 fishes) to the Large treatment total fish abundance.

Forty-two species were found to be unique to one of the four specific treatments. Of these 42 species, 26 (62%) were single fish counted one time over all the censuses, seven (17%) were a species counted twice, three (7%) were a species counted three times and four (10%) were species counted 4-10 times. Two species (4%) had single counts over ten: rainbow runner (*Elagatis bipinnulata*) with 60 fish on Quad-19 (Small treatment) during April 2003 and smallmouth grunt (*Haemulon chrysargyreum*) with 19 fish on Quad-16 (Empty treatment) during January 2004. Additionally, a single rock hind (*Epinephelus adscensionis*) was recorded on Quad-27 (Mixed treatment) during four successive counts and showed an increase in size over time (Oct. 2002, 5-10 cm; Jan. 2003, 5-10 cm; July 2003, 10-20 cm; Oct. 2003, 20-30 cm).

Table 1: List of fish species and abundance recorded from each of the reef ball treatments.

COMMON NAME	SCIENTIFIC NAME	Treatment: Empty	Small	Mixed	Large	Total
FAMILY: GUITARFISH	**RHINOBATIDAE**					
Atlantic Guitarfish	*Rhinobatos lentiginosus*				**1**	1
FAMILY: STINGRAY	**DASYATIDAE**					
Southern Stingray	*Dasyatis americana*		3		2	5
FAMILY: ROUNDRAYS	**UROLOPHIDAE**					
Yellow Stingray	*Urobatis jamaicensis*	7	3	6	8	24
FAMILY: REMORA	**ECHENEIDAE**					
Sharksucker	*Echeneis naucrates*		1	1	1	3
FAMILY: MORAY EELS	**MURAENIDAE**					
Chestnut Moray	*Enchelycore carychroa*	1	1	1	1	4
Goldentail Moray	*Gymnothorax miliaris*				**2**	2
Purplemouth Moray	*Gymnothorax vicinus*			1	5	6
FAMILY: HAWKFISH	**CIRRHITIDAE**					
Redspotted Hawkfish	*Amblycirrhitus pinos*		3	5		8
FAMILY: LIZARDFISHES	**SYNODONTIDAE**					
Inshore Lizardfish	*Synodus foetens*	1		1	2	4
Sand Diver	*Synodus intermedius*	**1**				1
FAMILY: SQUIRRELFISHES	**HOLOCENTRIDAE**					
Blackbar Soldierfish	*Myripristis jacobus*	1	1	5	9	16
Dusky Squirrelfish	*Sargocentron vexillarium*	**1**				1
Longspine Squirrelfish	*Holocentrus rufus*			**1**		1
Squirrelfish	*Holocentrus adscensionis*	2		15	25	42
FAMILY:BIGEYE	**PRIACANTHIDAE**					
Bigeye	*Priacanthus arenatus*				**1**	1
FAMILY: TRUMPETFISHES	**AULOSTOMIDAE**					
Trumpetfish	*Aulostomus maculatus*		**1**			1
FAMILY: TILEFISHES	**MALACANTHIDAE**					
Sand Tilefish	*Malacanthus plumieri*	1	3	1	4	9
FAMILY: SEA BASSES	**SERRANIDAE**					
Belted Sandfish	*Serranus subligarius*		**2**			2
Black Grouper	*Mycteroperca bonaci*			**1**		1

		Treatment				
COMMON NAME	**SCIENTIFIC NAME**	**Empty**	**Small**	**Mixed**	**Large**	**Total**
Butter Hamlet	*Hypoplectrus unicolor*			**1**		1
Coney	*Epinephelus fulvus*			**1**		1
Gag	*Mycteroperca microlepis*		1	1	1	3
Graysby	*Epinephelus cruentatus*	3	1			4
Greater Soapfish	*Rypticus saponaceus*		3	1		4
Harlequin Bass	*Serranus tigrinus*	4	3	2	4	13
Rock Hind	*Epinephelus adscensionis*			**4**		4
Sand Perch	*Diplectrum formosum*	195	116	133	119	563
Scamp	*Mycteroperca phenax*	1	6	2	3	12
Tobaccofish	*Serranus tabacarius*	**3**				3
Yellowmouth Grouper	*Mycteroperca interstitialis*		1		1	2
Grouper	*Mycteroperca sp.*		**1**			1
FAMILY: CARDINALFISHES	**APOGONIDAE**					
Barred Cardinalfish	*Apogon binotatus*			2	3	5
Flamefish	*Apogon maculatus*	12	5	9	33	59
Twospot Cardinalfish	*Apogon pseudomaculatus*	241	228	319	317	1105
Juvenile Apogonid	*Apogon sp.*		**1**			1
FAMILY: TUNAS	**SCOMBRIDAE**					
King Mackerel	*Scomberomorus cavalla*			**1**		1
Spanish Mackerel	*Scomberomorus maculatus*				**2**	2
FAMILY: JACKS	**CARANGIDAE**					
Almaco Jack	*Seriola rivoliana*	3		4	19	26
Amberjack	*Seriola dumerili*	27		11	16	54
Bar Jack	*Carangoides ruber*	27	24	6	6	63
Blue Runner	*Caranx crysos*	158	6	165	93	422
Rainbow Runner	*Elagatis bipinnulata*		**60**			60
Round Scad	*Decapterus punctatus*				**2**	2
Yellow Jack	*Carangoides bartholomaei*		4	1	1	6
Scad	*Decapterus sp.*	1	8	2	5	16
Juvenile Jacks	*Carangid sp.*	2		52		54
Juvenile Scad	*Decapterus sp.*				**3**	3
FAMILY: SNAPPERS	**LUTJANIDAE**					
Blackfin Snapper	*Lutjanus bucanella*	7		4	8	19

		Treatment				
COMMON NAME	**SCIENTIFIC NAME**	**Empty**	**Small**	**Mixed**	**Large**	**Total**
Cubera Snapper	*Lutjanus cyanopterus*			**1**		1
Gray Snapper	*Lutjanus griseus*		2	1	4	7
Lane Snapper	*Lutjanus synagris*				**7**	7
Mutton Snapper	*Lutjanus analis*	10	10	6	2	28
Yellowtail Snapper	*Ocyurus chrysurus*	14	26	27	29	96
FAMILY: MOJARRAS	**GERREIDAE**					
Yellowfin Mojarra	*Gerres cinereus*		**1**			1
FAMILY: GRUNTS	**HAEMULIDAE**					
Black Margate	*Anisotremus surinamensis*				**4**	4
Bluestripe Grunt	*Haemulon sciurus*	1	13	23	21	58
Caesar Grunt	*Haemulon carbonarium*		30		1	31
Cottonwick	*Haemulon melanurum*	28	76	165	292	561
French Grunt	*Haemulon flavolineatum*	5	41	92	80	218
Margate	*Haemulon album*		**1**			1
Porkfish	*Anisotremus virginicus*	52	54	68	68	242
Sailors Choice	*Haemulon parra*	1	9	20	16	46
Smallmouth Grunt	*Haemulon chrysargyreum*		**19**			19
Spanish Grunt	*Haemulon macrostomum*		26		7	33
Striped Grunt	*Haemulon striatum*		16	49	5	70
Tomtates	*Haemulon aurolineatum*	52	72	83	325	532
White Grunt	*Haemulon plumierii*	14	40	58	47	159
Juvenile Grunts	*Haemulon sp.*	311	1641	1167	1318	4437
FAMILY: PORGIES	**SPARIDAE**					
Grass Porgy	*Calamus arctifrons*	2	1	4	5	12
Jolthead Porgy	*Calamus bajonado*		1		2	3
Littlehead Porgy	*Calamus proridens*	5	8	4	7	24
Pluma	*Calamus pennatula*		5	4	1	10
Saucereye Porgy	*Calamus calamus*	30	27	47	14	118
Sheepshead Porgy	*Calamus penna*	5	11	5	5	26
FAMILY: DRUMS	**SCIAENIDAE**					
Cubbyu	*Equetus umbrosus*				**1**	1
Highhat	*Equetus acuminatus*	9	11	24	14	58
Jacknifefish	*Equetus lanceolatus*	21	9	10	16	56

COMMON NAME	SCIENTIFIC NAME	Treatment Empty	Small	Mixed	Large	Total
FAMILY: GOATFISHES	**MULLIDAE**					
Spotted Goatfish	*Pseudupeneus maculatus*	33	106	161	154	454
Yellow Goatfish	*Mulloidichthys martinicus*				**1**	1
FAMILY: SEA CHUBS	**KYPHOSIDAE**					
Bermuda Chub	*Kyphosus sectatrix*		**1**			1
FAMILY: BUTTERFLYFISHES	**CHAETODONTIDAE**					
Banded Butterflyfish	*Chaetodon striatus*	2	2	1		5
Foureye Butterflyfish	*Chaetodon capistratus*		**1**			1
Reef Butterflyfish	*Chaetodon sedentarius*	66	95	109	95	365
Spotfin Butterflyfish	*Chaetodon ocellatus*	8	16	22	9	55
FAMILY: ANGELFISHES	**POMACANTHIDAE**					
Blue Angelfish	*Holocanthus bermudensis*	6	12	10	8	36
French Angelfish	*Pomacanthus paru*	18	35	29	32	114
Gray Angelfish	*Pomacanthus arcuatus*	38	55	47	35	175
Queen Angelfish	*Holocanthus ciliaris*	10	23	17	22	72
Rock Beauty	*Holocanthus tricolor*	14	18	15	19	66
Townsend Angelfish	*Holocanthus sp.*		**2**			2
FAMILY: DAMSELFISHES	**POMACENTRIDAE**					
Beaugregory	*Stegastes leucostictus*	2	11	9	12	34
Bicolor Damselfish	*Stegastes partitus*	120	91	89	102	402
Blue Chromis	*Chromis cyanea*	1			6	7
Brown Chromis	*Chromis multilineata*	7	17	2	39	65
Cocoa Damselfish	*Stegastes variabilis*	4	6	17	15	42
Dusky Damselfish	*Stegastes fuscus*		2	4		6
Longfin Damselfish	*Stegastes diencaeus*		**1**			1
Purple Reeffish	*Chromis scotti*				**1**	1
Sergeant Major	*Abudefduf saxatilis*	2	3	3	24	32
Threespot Damselfish	*Stegastes planifrons*			1	4	5
FAMILY: WRASSES	**LABRIDAE**					
Blackear Wrasse	*Halichoeres poeyi*	4	3	2	8	17
Bluehead Wrasse	*Thalassoma bifasciatum*	1663	1957	1904	2106	7630
Bluelip Wrasse	*Cryptotomus roseus*		1		8	9
Clown Wrasse	*Halichoeres maculipinna*	16	35	32	65	148

COMMON NAME	SCIENTIFIC NAME	Treatment Empty	 Small	 Mixed	 Large	 Total
Green Razorfish	*Xyrichtys splendens*	12	2	1	4	19
Hogfish	*Lachnolaimus maximus*	38	45	60	77	220
Painted Wrasse	*Halichoeres caudalis*			**1**		1
Pearly Razorfish	*Xyrichtys novacula*	9	1	1		11
Puddingwife	*Halichoeres radiatus*	2	3		2	7
Rosy Razorfish	*Xyrichtys martinicensis*	11		1	4	16
Slippery Dick	*Halichoeres bivittatus*	887	790	849	744	3270
Spanish Hogfish	*Bodianus rufus*	3	2	5	2	12
Yellowcheek Wrasse	*Halichoeres cyanocephalus*			2	1	3
Yellowhead Wrasse	*Halichoeres garnoti*	15	15	19	59	108
Razorfish	*Xyrichtys sp.*		1	2		3
FAMILY: PARROTFISHES	**SCARIDAE**					
Bucktooth Parrotfish	*Sparisoma radians*	9	13	9	14	45
Greenblotch Parrotfish	*Sparisoma atomarium*	5	7	3	2	17
Princess Parrotfish	*Scarus taeniopterus*		1		4	5
Queen Parrotfish	*Scarus vetula*			**1**		1
Rainbow Parrotfish	*Scarus guacamaia*	2			3	5
Redband Parrot	*Sparisoma aurofrenatum*	33	90	80	43	246
Redfin Parrot	*Sparisoma rubripinne*	2	8	12	5	27
Redtail Parrotfish	*Sparisoma chrysopterum*	8	15	14	5	42
Stoplight Parrotfish	*Sparisoma viride*	12	20	18	16	66
Striped Parrotfish	*Scarus iseri*	2	5	4	5	16
Parrotfish	*Sparisoma sp.*	1		1	9	11
FAMILY: DRAGONETS	**CALLIONYMIDAE**					
Dragonet	Callionymidae	**1**				1
FAMILY: COMBTOOTH BLENNIES	**BLENNIDAE**					
Molly Miller	*Scartella cristata*	5	2	1	2	10
Redlip Blenny	*Ophioblennius atlanticus*	2	2		1	5
Seaweed Blenny	*Parablennius marmoreus*	65	53	87	64	269
Blenny	Blennidae	2	2		4	8
FAMILY: LABRISOMIDS	**LABRISOMIDAE**					
Hairy Blenny	*Labrisomus nuchipinnis*	2	2	1		5
Rosy Blenny	*Malacoctenus macrops*	1	4	3	5	13

		Treatment				
COMMON NAME	**SCIENTIFIC NAME**	**Empty**	**Small**	**Mixed**	**Large**	**Total**
Saddled Blenny	*Malacoctenus triangulatus*	13	20	14	11	58
FAMILY: CHAENOPSIDS	**CHAENOPSIDAE**					
Roughhead Blenny	*Acanthemblemaria aspera*		1	2		3
Sailfin Blenny	*Emblemaria pandionis*	**1**				1
FAMILY: GOBIES	**GOBIIDAE**					
Bridled Goby	*Coryphopterus glaucofraenum*	49	26	45	43	163
Colon Goby	*Coryphopterus dicrus*	1	5	1	3	10
Dash Goby	*Gobionellus saepepallens*	10	10	16	13	49
Goldspot Goby	*Gnatholepis thompsoni*	53	41	56	55	205
Neon Goby	*Gobiosoma oceanops*			3		3
Pallid Goby	*Coryphopterus eidolon*	**1**				1
Rusty Goby	*Priolepis hipoliti*				**2**	2
Seminole Goby	*Microgobius carri*	2	1			3
Glass/Masked Goby	*Coryphopterus hyalinus/personatus*	1	1			2
FAMILY: SURGEONFISHES	**ACANTHURIDAE**					
Blue Tang	*Acanthurus coeruleus*	32	71	99	102	304
Doctorfish	*Acanthurus chirurgus*	280	264	274	308	1126
Ocean Surgeon	*Acanthurus bahianus*	161	248	142	189	740
FAMILY: SCORPIONFISH	**SCORPAENIDAE**					
Spotted Scorpionfish	*Scorpaena plumieri*	1	3	5	2	11
FAMILY: LEFTEYE FLOUNDERS	**BOTHIDAE**					
Peacock Flounder	*Bothus lunatus*				**1**	1
FAMILY: LEATHERJACKETS	**BALISTIDAE**					
Gray Trigger	*Balistes capriscus*	96	113	140	117	466
Orange Filefish	*Aluterus schoepfi*			**6**		6
Orangespotted Filefish	*Cantherhines pullus*	17	8	13	12	50
Planehead Filefish	*Monocanthus hispidus*	52	67	64	64	247
Whitespotted Filefish	*Cantherhines macrocerus*	6	1	4	6	17
Filefish	*Aluterus sp.*	1	1			2
FAMILY: BOXFISHES	**OSTRACIIDAE**					
Honeycomb Cowfish	*Acanthostracion polygonia*	1	1	1		3
Scrawled Cowfish	*Acanthostracion quadricornis*	5	3	4	4	16
Smooth Trunkfish	*Lactrophrys triqueter*	4	2	4		10

COMMON NAME	SCIENTIFIC NAME	Treatment Empty	Small	Mixed	Large	Total
Spotted Trunkfish	*Lactrophrys trigonus*	**2**				2
Trunkfish	*Lactrophrys sp.*			**1**		1
FAMILY: PUFFERS	**TETRAODONTIDAE**					
Sharpnose Puffer	*Canthigaster rostrata*	104	102	101	87	394
FAMILY: SPINY PUFFERS	**DIODONTIDAE**					
Balloonfish	*Diodon holocanthus*	11	8	11	9	39
Porcupinefish	*Diodon hystrix*	5	5	5	3	18
Spotted Burrfish	*Chilomycterus atinga*	**3**				3
FAMILY: BROTULA	**BYTHITIDAE**					
Brotula	*Stygnobrotula sp.*	**1**				1
	Total Fishes	**5315**	**7214**	**7277**	**7859**	**27665**
	Total Species	**107**	**118**	**116**	**118**	**166**

3.1.1 Statistical comparisons – fish abundance

With fishes from all size classes combined into the four treatments (Empty, Small, Mixed, Large) there was a significant difference ($p \leq 0.0002$ ANOVA, TK) in the total fish abundances between the Empty treatment (48.3 ± 2.23 SEM) and the Small (65.7 ± 3.81), Mixed (66.2 ± 3.6), and Large (71.4 ± 3.3) while there were no significant differences between the remaining treatments ($p \geq 0.15$ ANOVA, TK) (Figure 13).

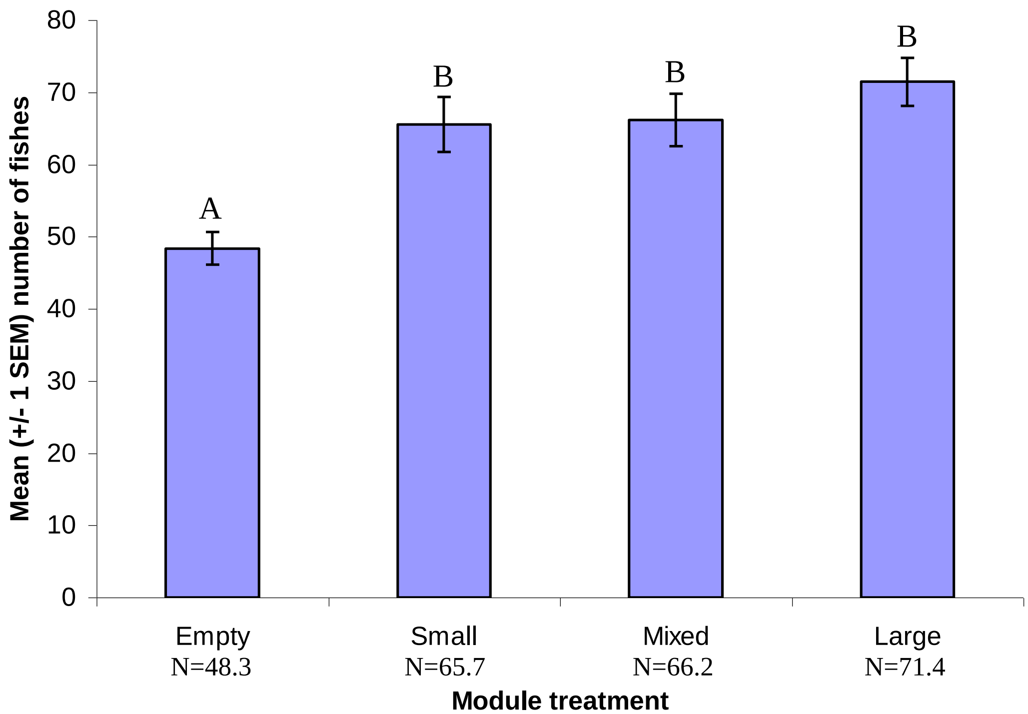

Figure 13: Mean abundances of fishes (± 1 SEM) counted within each treatment. Letters above each column indicate significant difference ($p < 0.05$ ANOVA, TK) between treatments.

Bluehead wrasse (15.1 ± 0.96 SEM) abundance showed a significant difference ($p = 0.04$, TK) only between the Empty and Large treatments (Figure 14). A significant difference existed for juvenile grunt abundance only between the Empty and other three treatments: Small (p = 0.0002, TK), Mixed (p = 0.0015, TK), and Large (p = 0.000, TK). There was no significant difference in the abundance of slippery dicks among any of the four treatments.

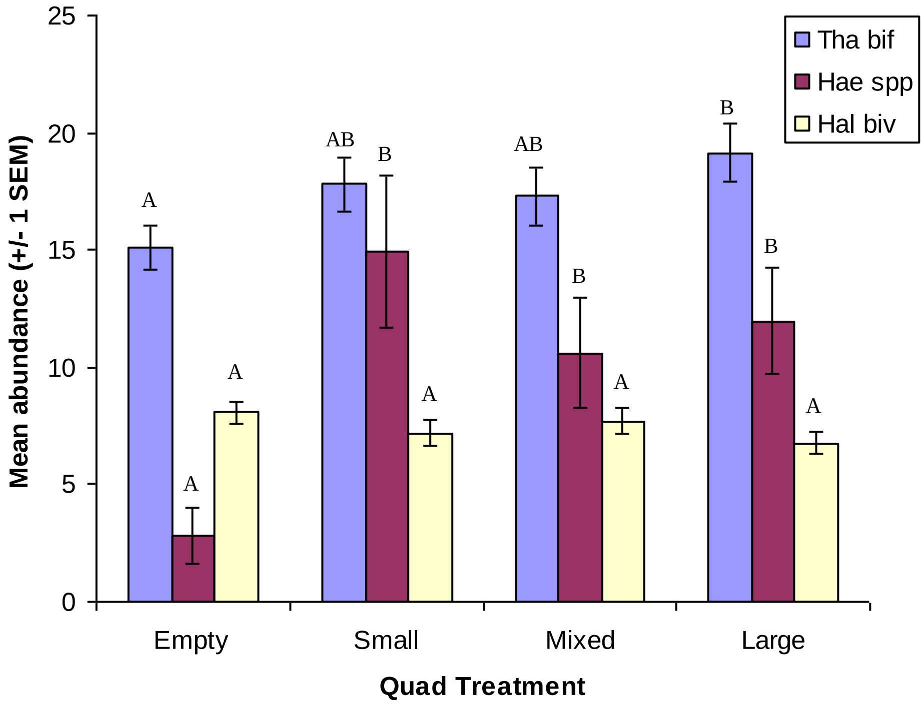

Figure 14: Mean abundance of bluehead wrasse (Tha bif), juvenile grunts (Hae spp), and slippery dicks (Hal biv) (± 1 SEM) within each treatment. Letters in each column indicate significant difference ($p < 0.05$ ANOVA, TK) between treatments within the species.

Examining the abundance data by size class, there was little difference between treatments for fish that were less than 2 cm (< 2 cm). Only Empty (3.1 ± 0.52) vs Mixed (5.5 ± 1.22) treatments showed a significant difference (p = 0.046 ANOVA, TK) (Figure 15). For the 2-5 cm size class, the Empty treatment (15.5 ± 1.4) was significantly different ($p \leq 0.0016$ ANOVA, TK) than the other three treatments (Small 25.3 ± 2.74; Mixed 24.0 ± 2.48; Large 27.8 ± 2.62) which did not show a significant difference compared to each other. Fishes in the 5-10 cm size class only showed a significant difference (p = 0.0094 ANOVA, TK) when the Empty treatment (16.0 ± 0.85) was compared to the Large treatment (22.4 ± 1.47). In the 10-20 cm size class, Empty (9.3 ± 0.79) was significantly different ($p \leq 0.01$ ANOVA, TK) than both Mixed (11.3 ± 0.65) and Large (11.6 ± 0.87) treatments while no other comparison showed any significant difference. The 20-30 cm size class again revealed a significant difference ($p \leq 0.008$ ANOVA, TK) when the Empty treatment (3.0 ± 0.81) was compared to both Mixed (5.1 ± 1.41) and Large (4.6 ± 0.68) treatments without there being any significant differences in the other treatment comparisons. Lastly, the size class of fishes greater than 30 cm (30^+ cm) did not show any significant difference between any of the four treatments.

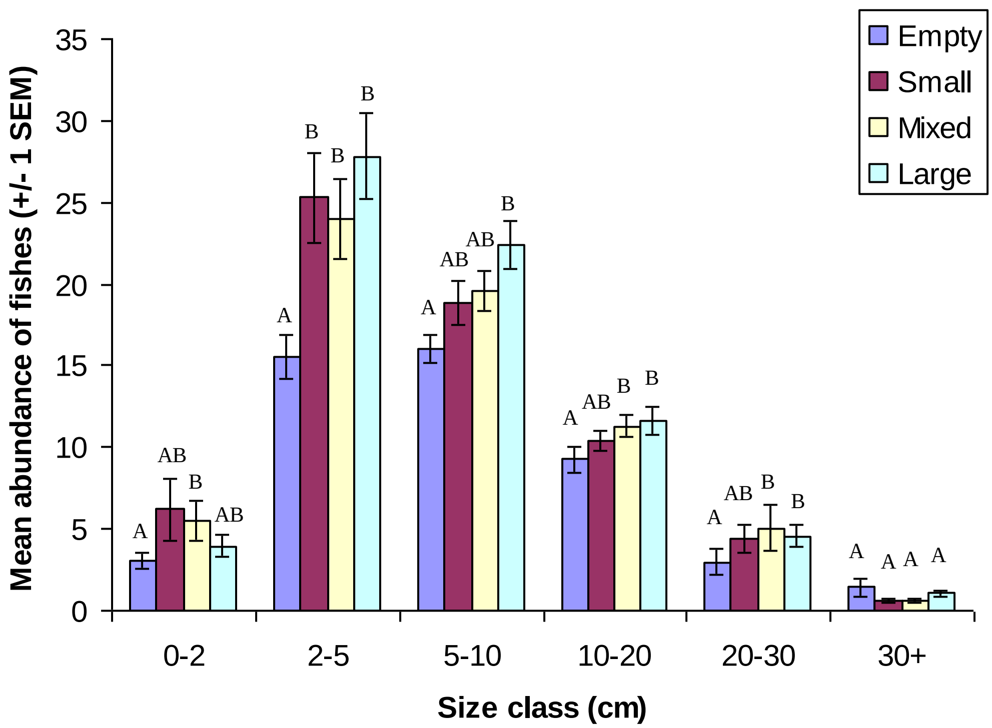

Figure 15: Mean abundance of fishes (± 1 SEM) by size class counted within each treatment. Letters in each column indicate significant difference ($p < 0.05$ ANOVA, TK) between treatments within the size class.

However, of the 155 fishes found on the Empty treatment in the 30^+ cm size class, 85 (55%) were blue runners (*Caranx crysos*) recorded during two censuses (April 2003 = 55 fish and January 2004 = 30 fish). While carangids were found in relatively high numbers on other treatments, the jacks represented a much smaller percentage (< 30%) of any size class.

3.1.2 Treatment comparisons – fish species richness

Similar to abundance results, when all size classes were pooled, species richness showed a highly significant difference ($p < 0.0001$ ANOVA, TK) between the Empty treatment (12.2 ± 0.3 SEM) and the other three treatments (Small 14.4 ± 0.2; Mixed 15.6 ± 0.3; Large 16.6 ± 0.3) (Figure 16). Additionally, there was a significant difference between the species richness of the Small vs Mixed treatments ($p = 0.03$ ANOVA, TK) and the Small vs Large treatments ($p = 0.001$ ANOVA, TK). The remaining treatment combination (Mixed vs Large) did not show a significant difference ($p = 0.54$ ANOVA, TK).

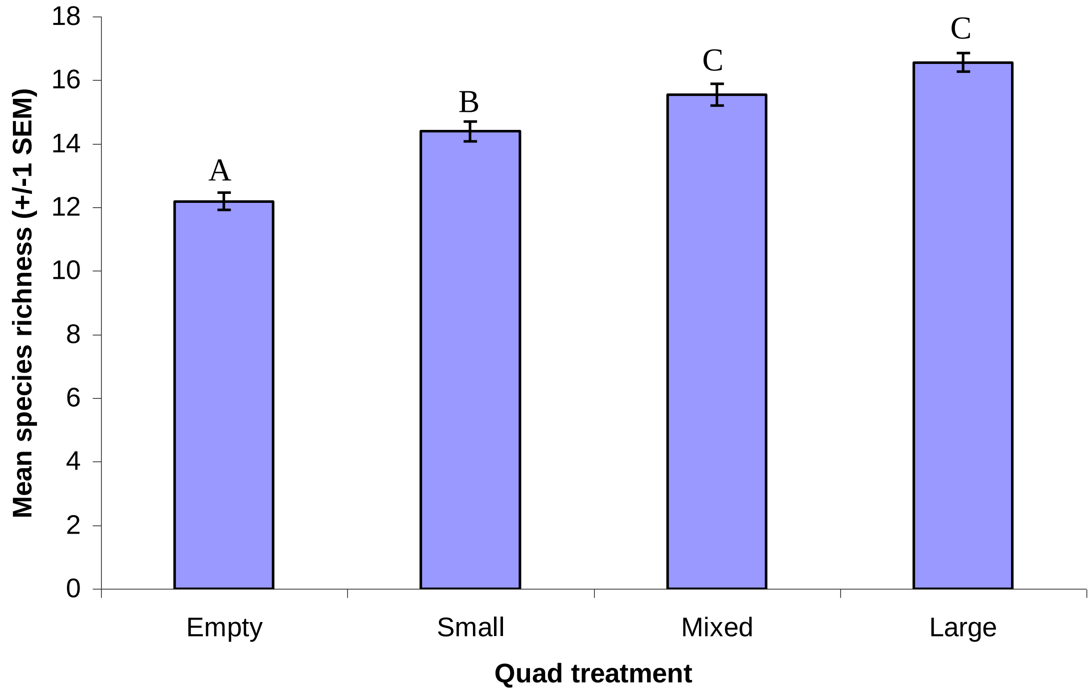

Figure 16: Mean species richness of fishes (± 1 SEM) counted within each treatment. Letters above each column indicate significant difference ($p < 0.05$ ANOVA, TK) between treatments.

When examining the richness data by size class, analysis of fishes in the 0-2 cm size class did not show a significant difference between any of the four treatments (Figure 17). For the 2-5 cm size class, only the Empty (4.4 ± 0.22) vs Large (5.1 ± 0.20) treatment showed a significant difference ($p = 0.04$ ANOVA, TK). The 5-10 cm size class showed a significant difference ($p \leq 0.011$ ANOVA, TK) between the Empty treatment (4.3 ± 0.18) and the other three treatments (Small 5.2 ± 0.20; Mixed 5.4 ± 0.21; Large 6.0 ± 0.22) in addition to a

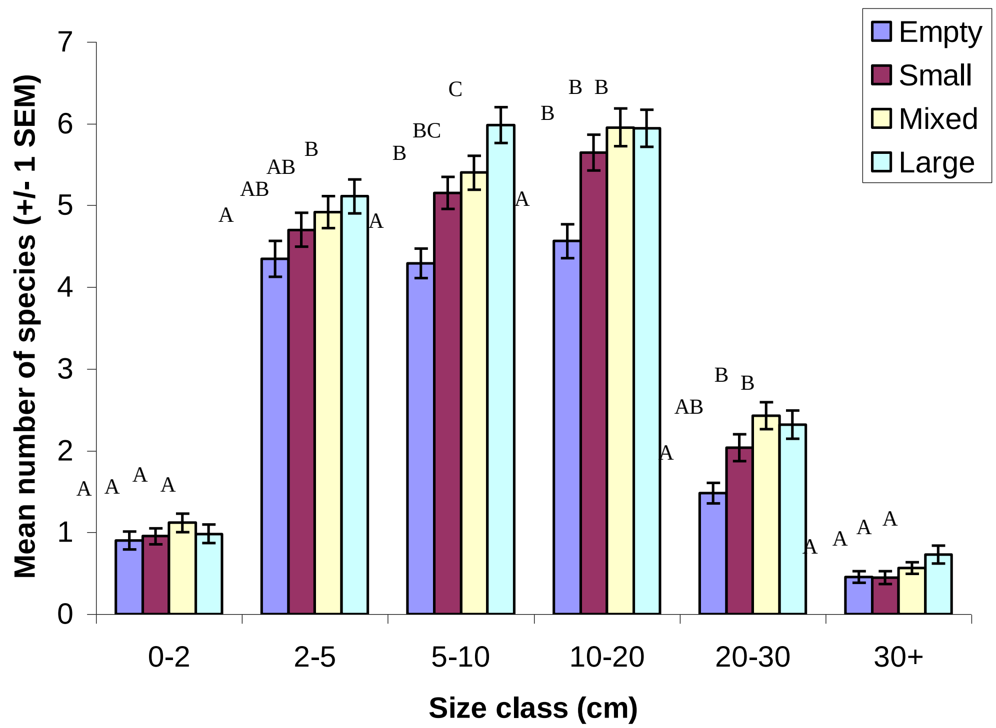

Figure 17: Mean species richness of fishes (± 1 SEM) by size class counted within each treatment. Letters above each column indicate significant difference ($p < 0.05$ ANOVA, TK) between treatments within the size class.

significant difference between the Large and Small treatments ($p = 0.016$ ANOVA, TK). The next size class, 10-20 cm, again showed a significant difference ($p \leq 0.006$ ANOVA, TK) between the Empty treatment (4.6 ± 0.21) and other three (Small 5.7 ± 0.0.22; Mixed 5.9 ± 0.0.23; Large 6.0 ± 0.23) while there was no significant difference between the Small, Mixed, or Large treatments. A significant difference ($p \leq 0.003$ ANOVA, TK) was only seen between the Empty (1.5 ± 0.13) vs Mixed (2.4 ± 0.17) and Empty vs Large (2.3 ± 0.17) treatments for the 20-30 cm size class. The final size class, 30^+ cm, did not show any significant difference in species richness between the four treatments.

3.1.3 Treatment comparisons – fish biomass

Overall, the Empty (2,360.10 g/quad ± 424.13 SEM), Small (2,442.04 ± 295.32), and Mixed (2,847.96 ± 450.87) treatments did not differ significantly for total mean biomass. The mean Large treatment biomass for all size classes combined (4,109.66 ± 1,232.35) was significantly greater compared to the other three treatments ($p \leq 0.012$ ANOVA, TK) (Figure 18).

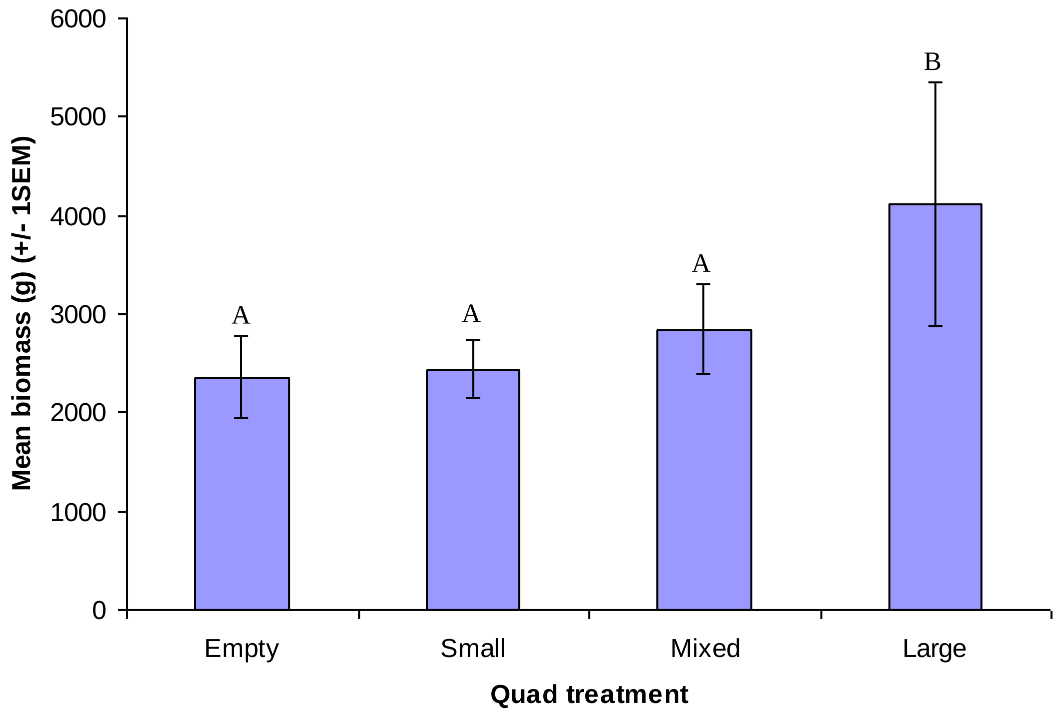

Figure 18: Total mean fish biomass by treatment (± 1 SEM). Letters above each column indicate significant difference ($p < 0.05$ ANOVA, TK) among treatments.

When analyzing biomass by size class, for all size classes except 0-2 cm and 20-30 cm, the Large treatment had a greater mean biomass than the other three treatments. The Small treatment had the greatest mean biomass (0.07 ± 0.02) of the 0-2 cm size class, while the Mixed treatment had the most biomass (1,693.39 ± 434.39) in the 20-30 cm size class.

The ($\log_{10}$ (x + 1)) transformation did not allow data in the 0-2 cm and 2-5 cm size classes to meet the assumptions of an ANOVA so these size classes were combined (0-5 cm) to create a larger data set for further analyses. Similarly, the 20-30 cm and 30^{+} cm size classes were combined (20-30^{+} cm).

The Empty treatment fish biomass for the 0-5 cm size class (8.80 ± 0.79) was significantly less ($p \leq 0.009$ ANOVA, TK) than the Small (15.57 ±1.82), Mixed (14.01 ± 1.53), and Large (16.56 ± 1.60) treatments. There were no significant differences between the other three treatments in this size class (Figure 19).

A significant difference ($p \leq 0.015$ ANOVA, TK) existed between the Empty (86.77 ± 5.12) vs Mixed (117.5 ± 8.74) and Empty vs Large (140.03 ± 11.20) in the 5-10 cm size class, however no other comparisons in this size class revealed a significant difference.

As with the 0-5 cm treatment comparisons, the 10-20 cm size class showed a significant difference ($p \leq 0.033$ ANOVA, TK) when the Empty treatment (579.20 ± 54.67) was compared to the remaining three treatments (Small 633.14 ± 34.64; Mixed 708.18 ± 39.32; Large 727.09 ± 62.27). Again, there was not a significant difference among the other three treatments.

In the final size class, 20-30^{+} cm, the Empty treatment had a less biomass (1,685.32 ± 425.0) and differed significantly ($p \leq 0.0023$ ANOVA, TK) from the Mixed (2,008.27 ± 435.27) and Large treatments (3,225.97 ± 1,235.26). Additionally, the Large vs Small (1685.53 ± 290.14) treatment comparison was also significantly different ($p = 0.038$ ANOVA, TK). The remaining comparisons, Empty vs Small, Small vs Mixed, and Mixed vs Large did not show a significant difference in biomass.

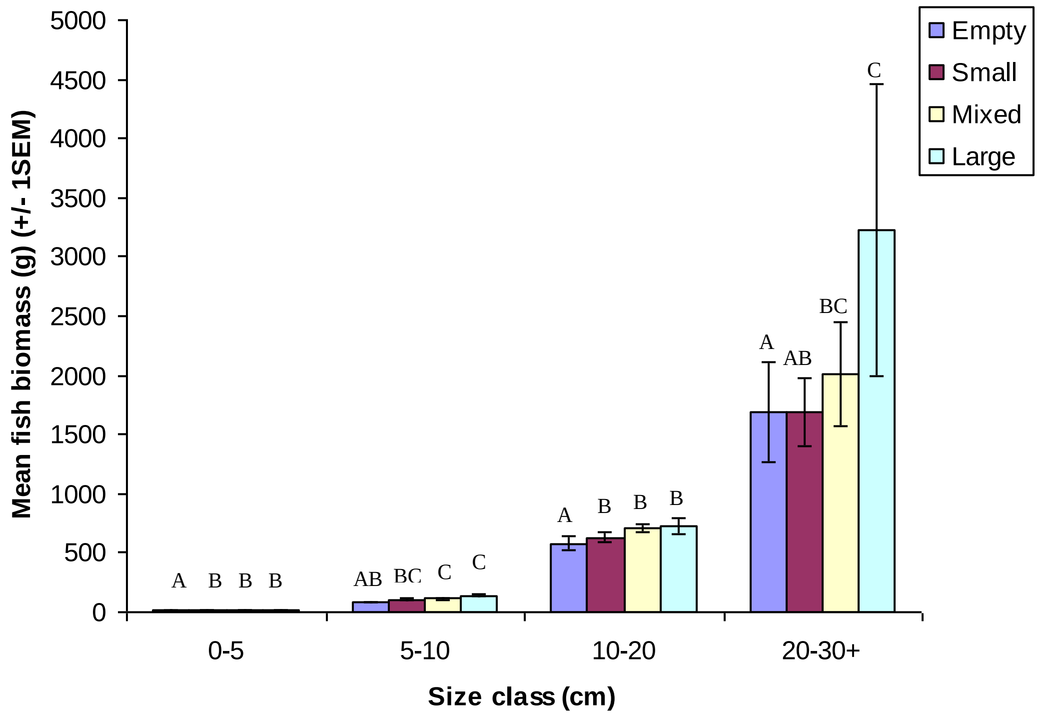

Figure 19: Mean fish biomass by size class (TL) and all size classes combined (± 1 SEM). Letters above each column indicate significant difference ($p < 0.05$ ANOVA, TK) between treatments within the size class.

3.1.4 Treatment comparisons – fish assemblage structure

Examination of the fish assemblage structure using multivariate analysis (MDS plot of Bray-Curtis similarity indices) revealed a significant difference regarding these fish assemblages on the four different treatments (Figure 20). Even with a high stress value (0.2), the difference between the Empty and Large treatments is apparent, although the 2-dimensional representation does not illustrate as clearly differences among other treatment comparisons. The Global R-statistic (ANOSIM) of 0.2 ($p = 0.001$) supported the MDS findings of significant differences among fish assemblages. An R-statistic value of 0 would indicate no difference while R=1 would be totally dissimilar assemblages. Empty treatment quads vs Large treatment quads had an R-statistic of 0.47 ($p = 0.002$, ANOSIM Pairwise test) which indicated significant difference with little overlap of fish assemblages.

ANOSIM Pairwise tests did not show a significant difference with the Small vs Mixed and Mixed vs Large treatments; however, the remaining treatment comparisons were all significantly different although R-statistic values were low (0.18 – 0.29) which again indicated overlap of the fish assemblages.

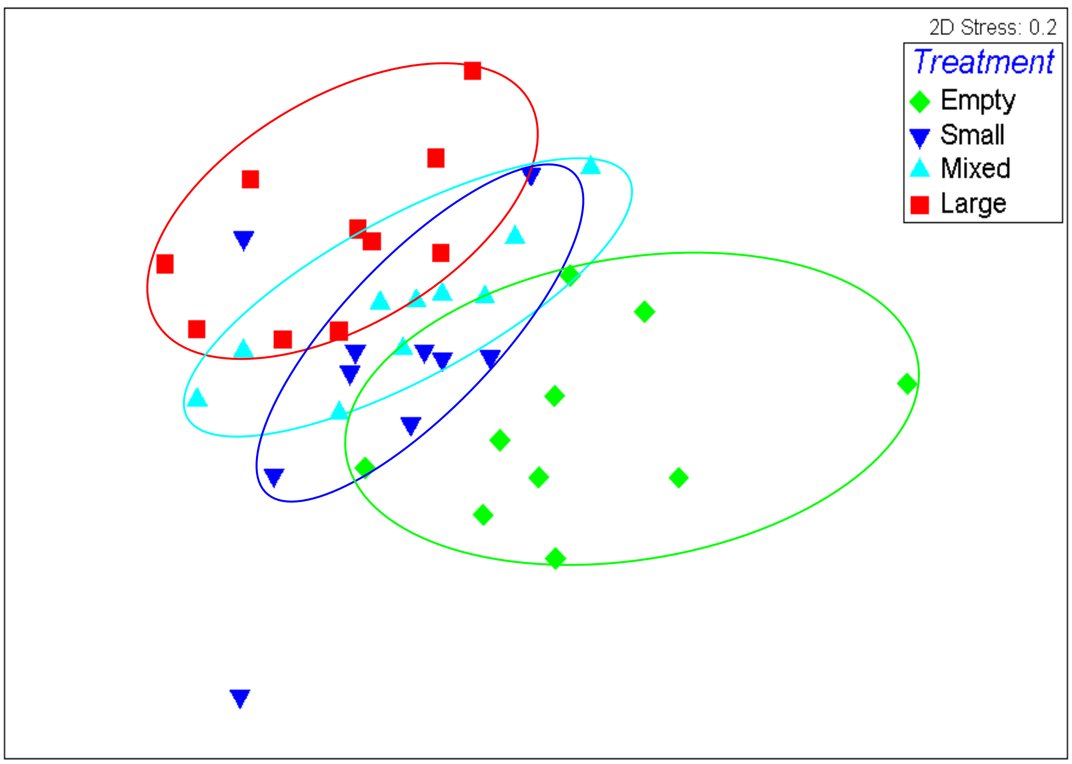

Figure 20: Fish species MDS plot of Bray-Curtis dissimilarity indices of individual treatments.

SIMPER analysis revealed which species contributed most to the differences indicated by the MDS plot (Table 2). The greatest dissimilarity between treatments existed between the Empty vs Large treatments (average dissimilarity 59.7%). Juvenile grunts contributed the greatest dissimilarity of 6.1%. The least dissimilarity was between Small vs

Mixed treatments with an average dissimilarity index of 57.5%. Again, juvenile grunts contributed the greatest amount to the dissimilarity with 7.3%.

Table 2: SIMPER percentages of the top ten species contributing most to the differences between the internal complexity treatments.

Empty vs Small Treatment (Average Dissimilarity = 58.0%)			
Scientific Name	Common Name	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Haemulon spp.	Juvenile Grunts	6.4	6.4
Acanthurus chirurgus	Doctorfish	4.6	11.0
Halichoeres bivittatus	Slippery Dick	4.5	15.5
Apogon pseudomaculatus	Twospot Cardinalfish	4.4	19.8
Acanthurus bahianus	Ocean Surgeonfish	4.2	24.1
Diplectrum formosum	Sand Perch	4.1	28.1
Thalassoma bifasciatum	Bluehead Wrasse	4.0	32.1
Stegastes partitus	Bicolor Damselfish	3.3	35.5
Balistes capriscus	Gray Trigger	3.2	38.7
Canthigaster rostrata	Sharpnose Puffer	3.1	41.8

Empty vs Mixed Treatment (Average Dissimilarity = 58.4%)			
Scientific Name	Common Name	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Haemulon spp.	Juvenile Grunts	5.7	5.7
Acanthurus chirurgus	Doctorfish	4.6	10.3
Apogon pseudomaculatus	Twospot Cardinalfish	4.4	14.7
Halichoeres bivittatus	Slippery Dick	4.2	18.9
Thalassoma bifasciatum	Bluehead Wrasse	4.1	23.0
Diplectrum formosum	Sand Perch	3.9	26.9
Acanthurus bahianus	Ocean Surgeonfish	3.7	30.5
Pseudupeneus maculatus	Spotted Goatfish	3.5	34.0
Balistes capriscus	Gray Trigger	3.3	37.3
Stegastes partitus	Bicolor Damselfish	3.1	40.4

Empty vs Large Treatment (Average Dissimilarity = 59.7%)			
Scientific Name	Common Name	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Haemulon spp.	Juvenile Grunts	6.1	6.1
Acanthurus chirurgus	Doctorfish	4.3	10.4
Apogon pseudomaculatus	Twospot Cardinalfish	4.2	14.6
Halichoeres bivittatus	Slippery Dick	4.0	18.7
Diplectrum formosum	Sand Perch	3.7	22.4
Acanthurus bahianus	Ocean Surgeonfish	3.6	26.00
Thalassoma bifasciatum	Bluehead Wrasse	3.6	29.5

Haemulon melanurum	Cottonwick	3.2	32.7
Pseudupeneus maculatus	Spotted Goatfish	3.0	35.7
Stegastes partitus	Bicolor Damselfish	3.0	38.7

Small vs Mixed Treatment (Average Dissimilarity = 57.5%)

Scientific Name	Common Name	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Haemulon spp.	Juvenile Grunts	7.3	7.3
Acanthurus chirurgus	Doctorfish	4.2	11.5
Apogon pseudomaculatus	Twospot Cardinalfish	4.2	15.7
Halichoeres bivittatus	Slippery Dick	4.2	19.9
Thalassoma bifasciatum	Bluehead Wrasse	3.9	23.8
Acanthurus bahianus	Ocean Surgeonfish	3.8	27.6
Diplectrum formosum	Sand Perch	3.2	30.8
Pseudupeneus maculatus	Spotted Goatfish	3.2	34.0
Balistes capriscus	Gray Trigger	3.2	37.2
Canthigaster rostrata	Sharpnose Puffer	2.9	40.1

Small vs Large Treatment (Average Dissimilarity = 58.4%)

Scientific Name	Common Name	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Haemulon spp.	Juvenile Grunts	7.3	7.39
Apogon pseudomaculatus	Twospot Cardinalfish	4.1	11.3
Halichoeres bivittatus	Slippery Dick	4.0	15.3
Acanthurus chirurgus	Doctorfish	4.0	19.3
Acanthurus bahianus	Ocean Surgeonfish	3.6	22.9
Thalassoma bifasciatum	Bluehead Wrasse	3.4	26.3
Haemulon melanurum	Cottonwick	3.2	29.5
Pseudupeneus maculatus	Spotted Goatfish	3.0	32.6
Diplectrum formosum	Sand Perch	3.0	35.6
Balistes capriscus	Gray Trigger	2.9	38.4

Mixed vs Large Treatment (Average Dissimilarity = 58.1%)

Scientific Name	Common Name	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Haemulon spp.	Juvenile Grunts	6.8	6.8
Apogon pseudomaculatus	Twospot Cardinalfish	4.1	10.9
Acanthurus chirurgus	Doctorfish	4.0	14.9
Halichoeres bivittatus	Slippery Dick	3.9	18.7
Thalassoma bifasciatum	Bluehead Wrasse	3.5	22.3
Haemulon melanurum	Cottonwick	3.5	25.8
Acanthurus bahianus	Ocean Surgeonfish	3.3	29.1
Pseudupeneus maculatus	Spotted Goatfish	3.2	32.3
Diplectrum formosum	Sand Perch	3.0	35.3
Balistes capriscus	Gray Trigger	2.9	38.3

3.2 Coral Recruitment

To determine coral recruitment, a single census of 80 modules (M2 and M3) was conducted at the end of the study. A total of 186 coral recruits were counted: 30 on the settlement plates and 156 on 80 artificial reef modules. One hundred thirty-seven recruits were able to be identified to the species level (nine species) and 47 to genus (five genera) while two were unable to be identified in-situ below the Order Scleractinia. Of the 320 settlement plates initially attached to the modules, 11 were lost or damaged over the course of the study. Three plates were from 2 Empty treatment quads, 2 plates were from 1 Small treatment quad, 4 plates were from 2 Mixed treatment quads, and 2 plates were from 2 Large treatment quads.

Porites astreoides was the most abundant recruit species with 89 (47.8%) of the total recruits. This percentage may be an underestimate as 23 recruits (12.4%) were identified only to the genus level, *Porites* spp. *Agaricia agaricites* was the second most abundant species with 25 (13.4%) recruits. This may also be an underestimate as 13 recruits (6.9%) were identified only the to level of *Agaricia* spp. The combined number of recruits from these two genera (154) accounted for 82.8% of the total number of recruits counted (Table 3).

Table 3: Species and number of coral recruits observed on quads by treatment.

Coral Species	Empty	Small	Mixed	Large	Total
Agaricia agaricites	4	3	12	6	**25**
Agaricia fragilis		1		1	**2**
Agaricia spp.	3	2	4	4	**13**
Diploria labyrinthiformis	1		1	2	**4**
Diploria strigosa	1	1	1	1	**4**
Diploria spp.	2	2	3	2	**9**
Meandrina meandrites	1		1	3	**5**
Phyllangia americana			5		**5**
Porities astreoides	9	34	27	19	**89**
Porities porities		1		1	**2**
Porities spp.	8	7		8	**23**
Siderastrea siderea		1		1	**2**
Siderastrea spp.				1	**1**
Scleractinia				2	**2**
Totals	**29**	**52**	**54**	**51**	**186**

3.2.1 Coral recruitment and quad treatment

With all coral recruits combined, there was a significant difference among the four treatments (Empty, Small, Mixed, Large) ($p < 0.05$ ANOVA). There was a highly significant difference in the recruit abundances between Empty (2.9 ± 0.38) vs Small (5.2 ± 0.92) treatments ($p = 0.02$ ANOVA, NK) and Empty vs Mixed (5.4 ± 0.88) treatments ($p = 0.03$ ANOVA, NK) (Figure 21). There was also a significant difference between the Empty vs Large (5.1 ± 0.55) treatments ($p = 0.052$ ANOVA, NK). The remaining treatment comparisons (Small vs Mixed; Small vs Large; Mixed vs Large) did not show a significant difference between recruit abundances.

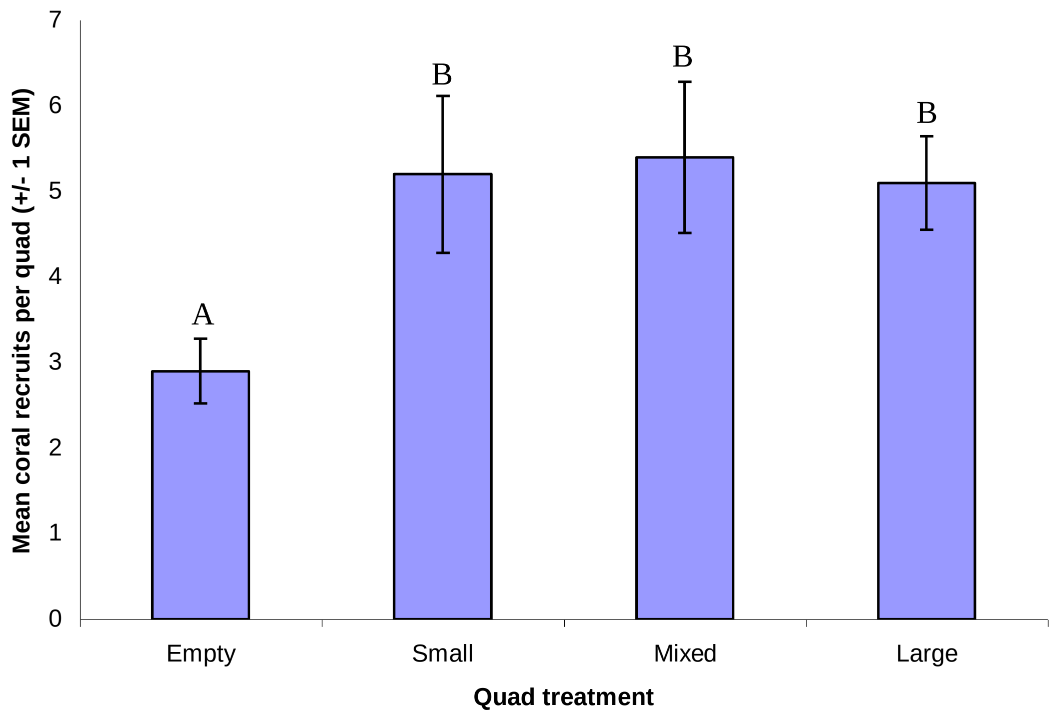

Figure 21: Mean coral recruit abundance (± 1 SEM) counted within each treatment. Letters in each column indicate significant difference ($p < 0.05$ ANOVA, NK) between treatments.

Empty treatment quads had the fewest number of recruits, a total of 29 (15.6%) in eight of the species/genera groups. *Porites astreoides* was the most abundant species with nine recruits followed by *Porites* spp. with eight recruits, which together made up 58.6% of the recruits found on the Empty treatment quads.

Small treatment quads had a total of 52 (28.0%) recruits in nine species/genera groups. Again, *P. astreoides* was the most abundant species with 34 recruits (65.4%) followed by *Porites* spp. with seven recruits (13.5%). Combined, these two groups comprised 78.9% of the total number of recruits found on the Small treatment quads.

Mixed treatment quads had the greatest number of recruits (54, 29%) in eight species/genera groups. *Porites astreoides* was the most abundant species with 27 recruits

(50%); however, *A. agaricites* was the second most abundant species with 12 recruits (22.2%). Together, these two species comprised 72.2% of the total number of recruits found on the Mixed treatment quads.

Large treatment quads had a total of 51 recruits (27.4%), but was the most speciose treatment with recruits from 14 (93.3%) of the species/genera groups identified in the study. As with the previous three treatments, *P. astreoides* was the most abundant species with 19 recruits (37.3%), followed by *Porites* spp. with eight (15.7%) recruits.

Among individual quads, Quad-4 (Small treatment) and Quad-6 (Mixed treatment) had the greatest number of recruits with 11 each. *Porites* spp. accounted for 81.8% (nine recruits) and 63.6% (seven recruits) respectively.

Due to the small number of recruits, abundance observations to the species level were combined and grouped by genus (*Agaricia, Diploria, Meandrina, Phyllangia, Porites*, or *Siderastrea*) for abundance comparison across treatments. However, there were no significant differences in abundance by genus among treatments when the data were grouped in this way.

Analyses of species richness did not show significant differences between Empty, Small, or Mixed treatments ($p > 0.05$ ANOVA, NK), but did show a significant difference between these three treatments when individually compared to the Large treatment ($p \leq 0.03$ ANOVA, NK) (Figure 22).

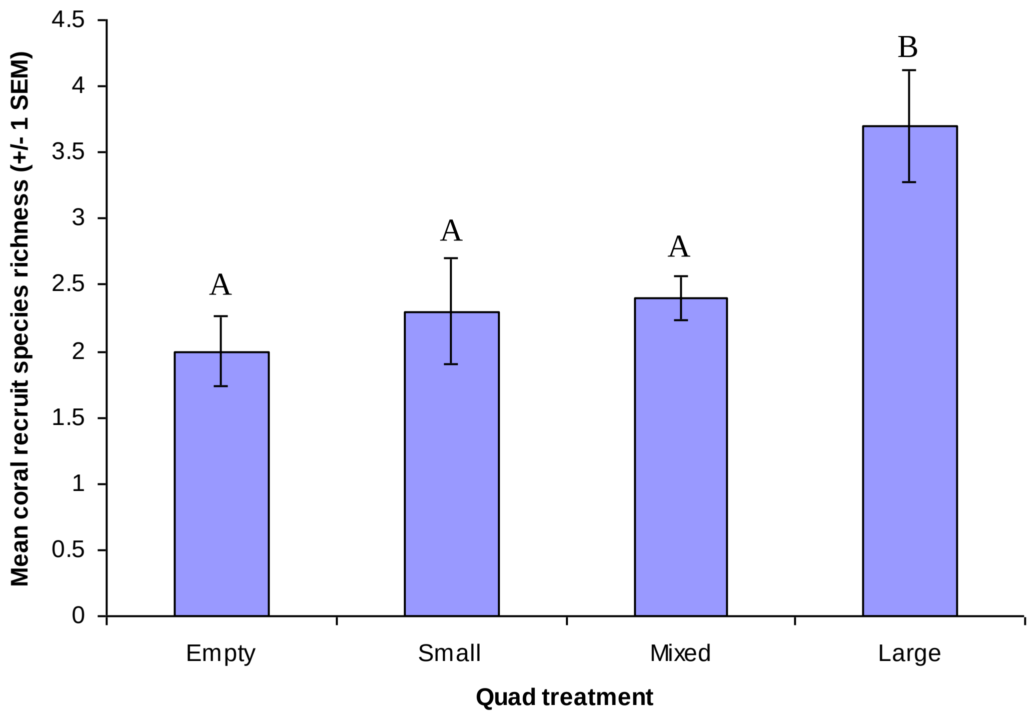

Figure 22: Mean species richness of coral recruits (± 1 SEM) counted within each treatment. Letters in each column indicate significant difference ($p < 0.05$ ANOVA, NK) between treatments.

Examination of the coral assemblage structure using multivariate analysis (MDS plot of Bray-Curtis similarity indices) did not reveal a significant difference regarding the recruit assemblages on the four different treatments (Figure 23). The Global R-statistic (ANOSIM) of 0.04 ($p = 0.15$) supported the MDS plot. Additionally, there were no pairwise comparisons between treatments for coral recruitment that showed a significant difference in the recruit assemblages.

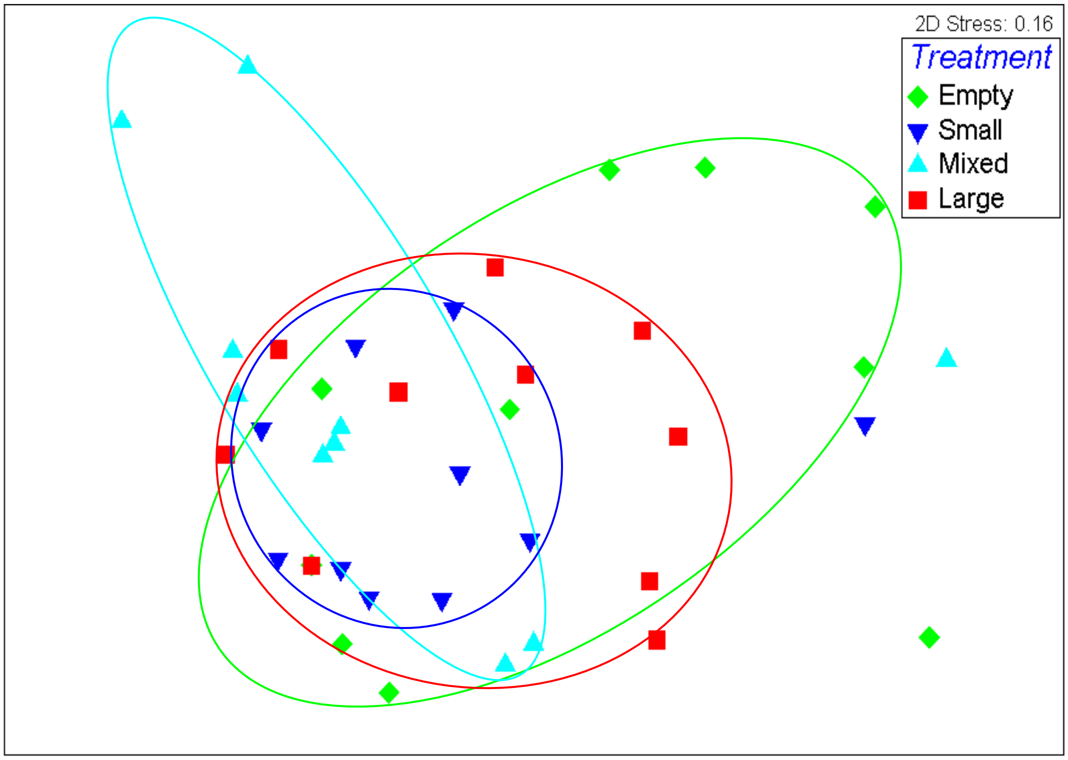

Figure 23: Coral recruits MDS plot of Bray-Curtis dissimilarity indices of individual treatments.

SIMPER analysis showed the greatest average dissimilarity between treatments existed between the Empty vs Mixed treatments (73.2%). *Porites astreoides* contributed the greatest dissimilarity at 25.6%. The least dissimilarity was between Small vs Large treatments with an average index of 56.3%. Again, *P. astreoides* contributed the greatest amount to the dissimilarity with 21.4%. When analyzed using the lowest identified taxonomic grouping, *Porites* and *Agaricia* recruits contributed the most (up to 73.9%, Empty vs Small treatments) to the dissimilarity between all possible combinations of treatments.

Table 4: SIMPER percentages of the top coral recruit species contributing most to the differences in recruit assemblages.

Empty vs Small (Average Dissimilarity = 67.1%)		
Species	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Porites astreoides	36.7	36.7
Porites spp	18.3	55.0
Agaricia agaricites	11.8	66.8
Agaricia spp	10.5	77.3
Diploria spp	8.0	85.2
Diploria strigosa	4.4	89.7
Agaricia fragilis	2.4	92.0

Empty vs Mixed (Average Dissimilarity = 73.2%)		
Species	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Porites astreoides	25.6	25.6
Agaricia agaricites	19.8	45.4
Porites spp	14.5	60.0
Agaricia spp	10.7	70.6
Diploria spp	8.9	79.5
Phylangia americana	8.2	87.7
Diploria strigosa	4.0	91.7

Empty vs Large (Average Dissimilarity = 66.8%)		
Species	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Porites astreoides	24.6	24.6
Porites spp	15.1	39.7
Agaricia agaricites	12.9	52.6
Agaricia spp	10.6	63.1
Diploria spp	7.0	70.1
Diploria labyrinthiformis	5.6	75.7
Scleractinia	4.4	80.1
Meandrina meandrites	4.3	84.4
Meandrina spp	4.3	88.7
Diploria strigosa	3.5	92.2

Small vs Mixed Treatment (Average Dissimilarity = 61.6%)		
Scientific Name	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Porites astreoides	27.0	27.0
Agaricia agaricites	20.1	47.1
Porites spp	12.6	59.6
Diploria spp	9.0	68.6
Phylangia americana	8.6	77.3

Agaricia spp	8.3	85.5
Diploria strigosa	3.9	89.4
Meandrina meandrites	2.8	92.3

Small vs Large Treatment (Average Dissimilarity = 56.3%)		
Species	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Porites astreoides	21.4	21.4
Porites spp	15.4	36.7
Agaricia agaricites	12.6	49.4
Agaricia spp	10.3	59.7
Diploria spp	7.1	66.8
Diploria labyrinthiformis	4.6	71.4
Scleractinia	4.6	76.0
Meandrina meandrites	4.6	80.6
Siderastrea siderea	3.8	84.4
Agaricia fragilis	3.6	87.9
Diploria strigosa	3.5	91.4

Mixed vs Large Treatment (Average Dissimilarity = 63.5%)		
Species	Contributed Dissimilarity (%)	Cumulative Dissimilarity (%)
Porites astreoides	19.9	19.9
Agaricia agaricites	15.9	35.8
Porites spp	11.6	47.4
Agaricia spp	10.7	58.1
Diploria spp	7.7	65.7
Phylangia americana	7.4	73.2
Meandrina meandrites	5.3	78.5
Diploria labyrinthiformis	5.1	83.5
Scleractinia	3.9	87.4
Diploria strigosa	3.1	90.5

Size classes of 1-12, 13-25, 26-38, 39-51, 52-64, and 65-77 mm were established for the coral recruits using a conservative growth rate of 12 mm yr^{-1} based on previous reports (Edmunds et al. 2004) and the assumption that recruits grow at a similar rate (van Moorsel 1988). Recruit size ranged from 3 mm diameter (*Phyllangia*) to 66 mm diameter (*Porites*) (Table 5). One hundred seventy-nine recruits (96%) were ≤ 38 mm in diameter with *Porites* accounting for 61% (109 recruits) and *Agaricia* 21.2% (38 recruits) (Figure 24). *Porites* (7 recruits) and *Agaricia* (5 recruits) were the only taxa in the larger size classes (≥ 39 mm).

Table 5: Number of corals recruited to the artificial reef modules by taxa and size class (mm). Size classes determined by conservative growth rate estimate of 12 mm/yr^{-1}.

Diameter (mm)	*Agaricia*	*Diploria*	*Meandrina*	*Phyllangia*	*Porites*	*Siderastrea*	Scleractinia	Total
1-12	6	2	2	4	38	1	1	**54**
13-25	22	10	2		49	1	1	**85**
26-38	7	5	1	1	20	1		**35**
39-51	4				4			**8**
52-64	1				2			**3**
65-77					1			**1**
Total	**40**	**17**	**5**	**5**	**114**	**3**	**2**	**186**

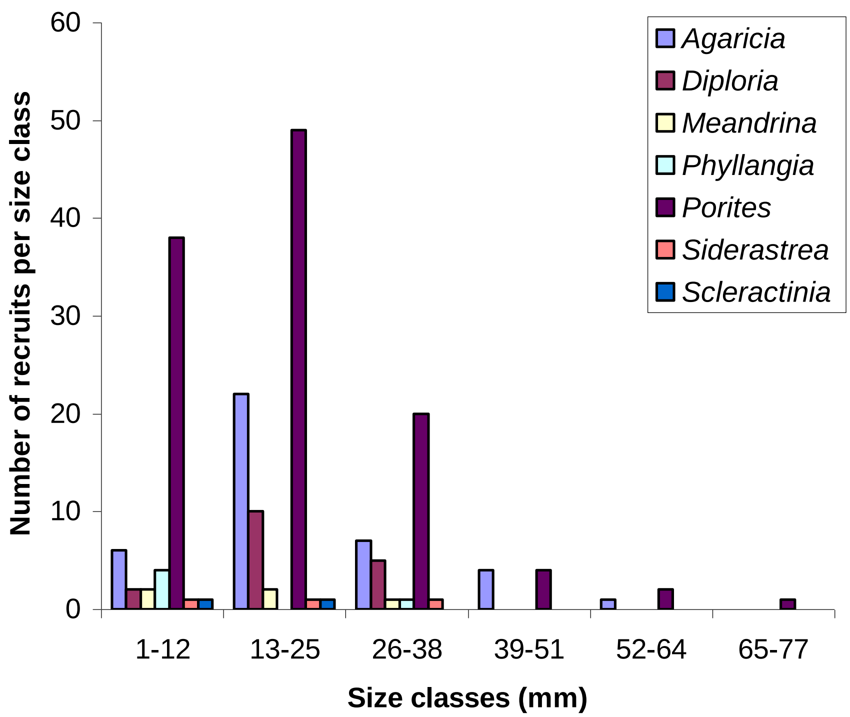

Figure 24: Number of coral recruits grouped by diameter size class (mm). Size classes determined by conservative growth rate estimate of 12 mm/yr^{-1}.

To examine differences in recruit size, data were grouped by taxa to create larger sample sizes for an analysis of variance. Even with the larger sample size, there were no significant differences between the 7 taxa for mean recruit size (Figure 25).

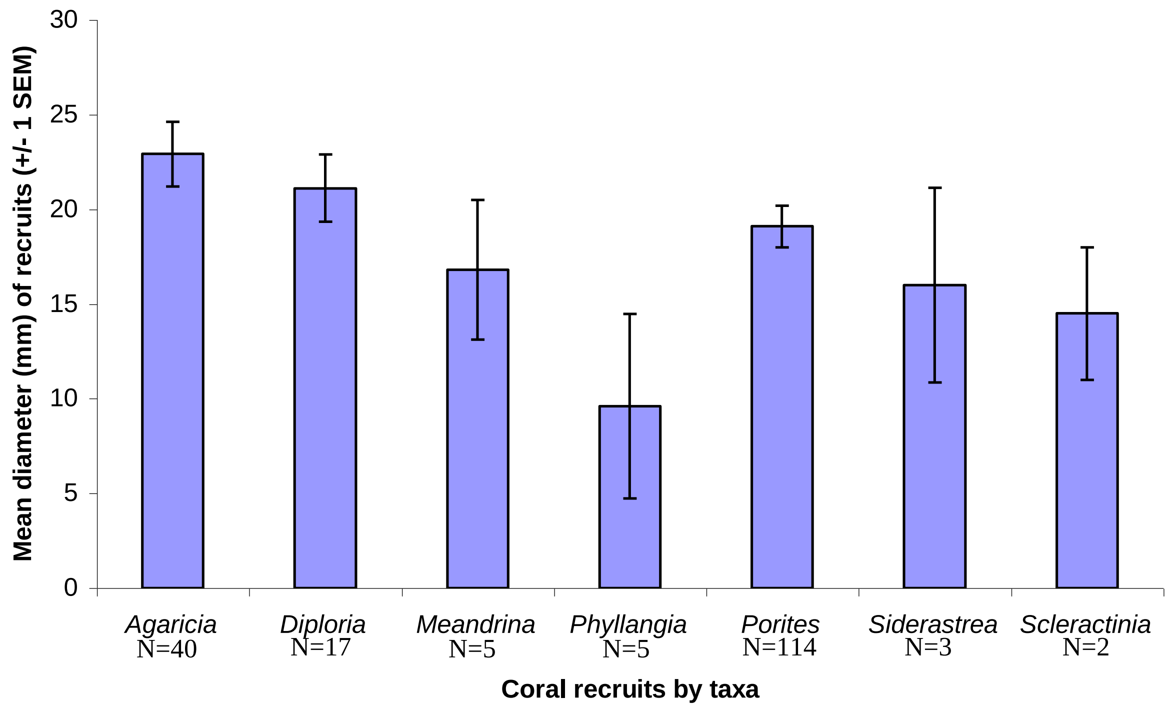

Figure 25: Mean coral recruit diameter (mm) (± 1 SEM) grouped by taxa. There were no significant differences between recruit groups ($p > 0.05$ ANOVA).

3.2.2 Coral recruitment/fish assemblage correlations

Correlation analyses were conducted using coral recruit data in the size classes reported above (Table 5) with the exception that the three largest size classes, 39-51 mm, 52-64 mm, and 65-77 mm, were combined and considered to have settled in Year 1 after deployment. The 26-38 mm size class was considered to have settled in Year 2 after deployment, the 13-25 mm size class in Year 3 after deployment, and recruits in the 1-12 mm size class considered to have recruited in the final months (Year 4) of the study.

All fish census data were partitioned by year, with all 11 counts (Oct. 2001 – May 2004) pooled and correlated against both the Year 1 recruit size class ($n = 12$) and all coral recruits combined; 10 counts (Jan. 2002 – May 2004) were pooled and correlated against the Year 2 recruit size class ($n = 35$); 6 counts (Jan. 2003 – May 2004) were pooled and correlated against the Year 3 size class ($n = 85$); 2 counts (Jan. 2004 – May 2004) were pooled and correlated against the Year 4 size class ($n = 54$).

Neither standard or transformed data showed a significant correlation ($p > 0.05$) with parametric and non-parametric tests between any of the coral recruit size classes and the corresponding fish assemblages by year or between total fish and recruits. Similar analyses were conducted using all of the coral recruits, but with only the top 90% (26 species) of fish (24,914) by abundance, however no significant correlation was apparent.

Correlation analyses were also run using only butterflyfishes, damselfishes, grunts, parrotfishes, pufferfishes, surgeonfishes, triggerfishes, wrasses, and the selected families combined. These families totaled 82.4% of the total fish counted or 22,799 fish in 62 species. Damselfish were the only taxa to show a significant (parametric) correlation ($r = 0.32$, $p < 0.05$) with coral recruits in Year 1 (Table 6). No taxa showed a correlation in Year 2. Grunts ($r = 0.37$, $p < 0.05$, Spearman Rank) and all families combined ($r = 0.41$, $p < 0.05$, Spearman Rank) showed a significant correlation in Year 3. In Year 4, only butterflyfish showed a correlation ($r = 0.39$, $p < 0.05$, Spearman Rank).

Table 6: Coral recruits by size class and fish select-species correlated individually and all select-species combined. Numbers in bold text indicate significant correlation ($p < 0.05$).

Coral recruits / Fishes	Year 1 (39-51+ mm) n = 12	Year 2 (26-38 mm) n = 35	Year 3 (13-25 mm) n = 85	Year 4 (1-12 mm) n = 54
butterflyfishes	0.05 n = 426	0.14 n = 393	0.13 n = 253	**0.39** n = 88
damselfishes	**0.32** n = 595	-0.09 n = 573	0.04 n = 332	0.01 n = 85
grunts	0.14 n = 6411	-0.10 n = 5748	**0.37** n = 3397	0.27 n = 1717
parrotfishes	-0.16 n = 481	0.05 n = 448	0.06 n = 316	0.28 n = 111
pufferfishes	-0.05 n = 454	0.00 n = 425	-0.11 n = 200	-0.16 n = 33
surgeonfishes	0.09 n = 2170	0.02 n = 2060	0.21 n = 1048	0.03 n = 273
triggerfishes	0.21 n = 788	-0.04 n = 702	0.18 n = 359	0.29 n = 165
wrasses	-0.05 n = 11474	-0.15 n = 10560	0.06 n = 7008	-0.15 n = 3220
families combined	0.17 n = 22799	-0.09 n = 20909	**0.41** n = 13513	0.27 n = 5692

A further correlation analysis was completed with these families of fishes, but only with those species found in the top 90% by abundance. Only the reef butterflyfish (*Chaetodon sedentarius*), bicolor damselfish (*Stegastes partitus*), redband parrotfish (*Sparisoma aurofrenatum*), sharpnose puffer (*Canthigastor rostata*), grey trigger (*Balistes capriscus*), and planehead filefish (*Monocanthus hispidus*) were most abundant of their respective families. Species of grunts were cottonwicks (*Haemulon melanurum*), tomtates (*H. aurolineatum*), french grunts (*H. flavolineatum*), white grunts (*H. plumerii*), porkfish (*Anisotrimus virginicus*), and juvenile grunts. All three surgeonfishes: blue tang (*Acanthurus coeruleus*), doctorfish (*A. chirurgus*), and surgeonfish (*A. bahianus*) were represented in the top 90% of these selected species. Lastly, three species of wrasses, bluehead wrasse

(*Thalasomma bifasciatum*), slippery dick (*Halichoeres bivittatus*), and hogfish (*Lachnolaimus maximus*) were included. These species totaled 21,559 fish, 77.9% of the total number of fish counted.

For Year 1 correlation, only the reef butterflyfish showed a significant relationship with coral recruits ($r = 0.34$, $p < 0.05$, Spearman Rank) (Table 7). There were no significant correlations in Year 2. Year 3 showed a significant parametric correlation with the analysis of grunts ($r = 0.31$, $p < 0.05$) and all families combined ($r = 0.31$, $p < 0.05$). The Year 4 analysis again showed a significant correlation with reef butterflyfish ($r = 0.38$, $p < 0.05$, Spearman Rank).

Table 7: Coral recruits by size class and fish select-species in the top 90% by abundance correlated individually and all select-species combined. Numbers in bold text indicate significant correlation ($p < 0.05$).

Coral recruits / Fishes	Year 1 (39-51+ mm) n =12	Year 2 (26-38 mm) n =35	Year 3 (13-25 mm) n =85	Year 4 (1-12 mm) n =54
reef butterflyfish	**0.34** n = 365	0.21 n = 338	0.11 n = 217	**0.38** n = 76
bicolor damselfish	0.05 n = 402	0.01 n = 389	-0.07 n = 230	0.12 n = 50
grunts	0.19 n = 6149	-0.11 n = 5488	**0.98** n = 3162	0.30 n = 1607
redband parrotfish	0.09 n = 246	0.18 n = 230	0.13 n = 153	0.10 n = 64
sharpnose puffer	-0.05 n = 394	0.06 n = 374	0.10 n = 175	-0.15 n = 26
surgeonfishes	0.19 n = 2170	0.02 n = 2060	0.21 n = 1048	0.03 n = 273
triggerfishes	-0.16 n = 713	0.29 n = 639	0.00 n = 329	-0.05 n = 158
wrasses	-0.04 n = 11120	-0.11 n = 10262	0.18 n = 6902	0.21 n = 3194
families combined	0.21 n = 21559	-0.09 n = 19780	**0.38** n = 12216	0.27 n = 5448

3.2.3 Coral recruitment to settlement plates

Eleven of the 320 settlement plates were damaged or lost over the course of the study and thus were not included in the coral recruit assessment. Of these eleven plates, 3 were $CaCO_3$ treatment, 3 were iron treatment, 2 were transplant treatment and 3 were controls.

All 40 *Montastrea cavernosa* transplants survived the length of the study with no tissue die-off and growth ranging from 7 – 219% of the original transplant size. *Meandrina meandrites* transplants suffered some degree of mortality in 73% (29) of the transplants. Fifteen (38%) *M. meandrites* transplants experienced tissue loss (partial mortality) of 20 – 95% while an additional 14 transplants (35%) suffered total mortality. Colonies that suffered mortality were distributed amongst the treatment types: Empty – 3 partial mortality and 3 total mortality; Small – 6 partial and 3 total; Mixed – 1 partial and 5 total; Large – 5 partial and 3 total.

A total of 30 recruits were observed on the settlement plates: 11 on $CaCO_3$ treated plates, 6 on plates treated with iron, 6 on transplant plates, and 7 on controls. *Porites astreoides* accounted for 63.3% (19) of the total number of recruits with the remaining 36.7% (11) spread among 8 species/genera groups (Table 8).

Due to the small sample size, statistical analyses used in previous aspects of this study could not be used, however a Bray-Curtis Similarity Presence/Absence analysis (Primer v.6) was conducted on community structure among the four treatments. The $CaCO_3$ treatment had a 67% similarity in community structure with iron treatment, a 44% similarity with transplant treatment and only a 40% similarity with the control. The iron treatment had

Table 8: Species and number of coral recruits observed on settlement plates attached to artificial reef modules.

Coral Species	**$CaCO_3$**	**Iron**	**Transplant**	**Control**	**Totals**
Porities astreoides	7	2	5	5	**19**
Porities sp.				1	**1**
Diploria strigosa	1	1			**2**
Diploria sp.		1			**1**
Meandrina spp.			1	1	**2**
Agaricia agaricites	1				**1**
Agaricia spp.	1	1			**2**
Siderastrea sp.	1				**1**
Scleractinia		1			**1**
Totals	**11**	**6**	**6**	**7**	**30**

a 44% similarity with the transplant treatment and a 40% similarity with the control. The transplant treatment had an 86% similarity in community structure with the control due to *P. astreoides* accounting for most of the recruits found on these settlement plates.

4. Discussion

4.1 Effect of varying complexity on fish assemblages

Many studies, which used artificial reefs to examine how refuge complexity structures fish assemblages, have employed either a single type of refuge construct or a non-standardized refuge structure without replicates (Bohnsack 1983; Shulman 1985a, b; Hixon and Beets 1989, 1993; Gilliam 1999; Sherman et al. 2001; Brickhill et al. 2005). Reef fish are known to show strong site fidelity among patch reefs at least 10 m^2 in size (Sale et al. 1984) with negligible post-settlement movement (Caley et al. 1996). Clearly, the scale of an artificial reef must be large enough to develop a stable assemblage structure.

Each quad used in this study comprised a total artificial reef size of approximately 13 m^3. A total of 166 species were recorded, which is substantially higher than those reported in other artificial reef studies off Broward County. Gilliam (1999) reported 89 species on 40 small 1 m^3 layer-cake artificial reefs in 7 m depth, while Sherman (2000) reported 97 species on the same type of layer-cake reefs, but deployed at a depth of 21 m. Additionally, Sherman et al. (1999) reported 88 species on 16 modules, similar to the ones use in this study, with eight deployed in 7 m depth and eight deployed in 21 m depth. Caution should be used, however, when comparing results from these multiple studies. The modules used in Gilliam's (1999) and Sherman's (1999, 2000) studies were single modules, fewer modules were used than in my study, and were censused monthly for less than two years. Additionally, while counting techniques were identical, the total area for an individual quad censused in this study was estimated at 50 m^3, while the other studies' census area per module was 18 m^3. Although assemblages and census techniques were different, the 166 species counted on the quads more closely matched the richness (163 spp.) found in a study on larger artificial (vessel) reefs off Broward County (Arena 2005).

Jordan et al. (2005), using the same artificial reef modules as Gilliam (1999) and Sherman (2000), but placed at varying distances from each other to examine space effects, counted 139 species over two years of monthly censuses. Jordan (2005) suggested that modules placed close together (0.33 m vs 5 m in his study), similar to the quads in this study, performed as a larger individual reef rather than individual closely spaced modules. Freeman (2007) found 160 species on large boulder reefs deployed as mitigation for Broward County's beach nourishment although these artificial reefs were placed in the nearshore (5 m depth) environment. Bohnsack et al. (1994) reported 127 species on 2.3 m^3 grouped reefs in the Florida Keys. Eklund (1996) recorded 126 species using the same reefs in the Keys as Bohnsack, and 151 species on larger pyramidal reefs off Palm Beach County, located north of this study.

Studies on the natural reef environment in Broward County have recorded species numbers ranging from 151 species (Freeman 2007) on the nearshore hard-bottom to 208 species (Ferro et al. 2005) on the three parallel reef tracts. While much less than the number of species Ferro et al. (2005) reported, the total species reported in this study, compared with others, indicate that larger artificial reefs (e.g. multiple modules placed together or vessel artificial reefs) allow for a more diverse fish assemblage than smaller more isolated artificial reefs (single modules with separation distances of tens of meters). Thus size is an important consideration in using ARs in restoration efforts, mitigation projects, or to examine ecological processes occurring on the natural reefs.

When comparing abundance, species richness, and biomass across the four treatments, the Empty treatment was almost always significantly lower than the Small, Mixed, and Large treatments. The only exceptions were the Empty treatment did not differ from the other three treatments in abundance in the 30^+ cm size class and species richness in both the 0-2 cm and 30^+ cm size classes (Figures 15 and 17).

With all counts combined, the Empty treatment only totaled 311 juvenile grunts (the most abundant taxa in this study) while the Small had 1,641, Mixed had 1,167, and Large had 1,318 (Table 1) which was a significant difference between the Empty and other three treatments (Figure 15). Juvenile grunt totals were approximately 20 – 40% less in the Mixed and Large treatments when compared to the Small. Even though the Mixed treatment had two modules filled with cage material and one with block, one module did not contain any fill complexity which perhaps contributed to a concomitant reduction in juvenile grunts through predation although this was not a significant difference. The Large treatment quads had fill in each module and, while not providing the degree of small-opening refuge found in the Small treatment quads, apparently did provide sufficient predator-avoidance for significant juvenile grunt recruitment.

Bluehead wrasse (*Thalassoma bifasciatum*) and slippery dicks (*Halichoeres bivittatus*), the next two most abundant species, showed little individual difference between the four treatments with the exception that numbers of bluehead wrasse were significantly different between Empty and Small treatments. Small treatment quads provided more specific refuge from predators due to the small-sized openings in the cage material. Shima (2002), using a similar cage technique in Moorea, French Polynesia, found a significant decrease in the number of six bar wrasses (*Thalassoma hardwicke*) presumably lost to predation on uncaged reefs. In the Caribbean, Caselle and Warner (1996) found that microhabitat explained 30-35% of recruitment variances although they were unable to determine if settlement, movement to other reefs, or mortality was the cause. Roberts and Ormond (1987) found that the number of holes on a reef accounted for 77% of the fish abundance variation in the Red Sea, but this amount did vary by species and family and only fish with strong site attachment showed a positive relationship to the substratum's structural complexity. This contrasts with a study on the Great Barrier Reef by Caley and St. John (1996) who found total abundance and species richness of newly settled fishes did not differ significantly among shelter treatments for small artificial reefs and so determined there was no apparent habitat selection by fishes in response to differences in refuge availability. Another factor contributing to the lower numbers of bluehead wrasse on Empty treatment quads may be in part due to enhanced area epifaunal growth suitable for foraging provided by the internal substrate of the Small treatment (Eklund 1996).

Predation of new fish recruits is thought to occur within a few days after settlement (Doherty and William 1988; Bohnsack et al. 1994; Caley et al. 1996). Little difference was noted between the treatments in the 0-2 cm size class which may indicate fish settled indiscriminately on substrate and experienced post-settlement mortality that structured the assemblage. Eklund (1996) showed that fish in the 0-2 cm size class experienced equal mortality over reefs that were open to predators and reefs where predators were excluded. As discussed above, juvenile grunt abundance on the Empty treatment quads was significantly less than the other three treatments, while sand perch (*Diplectrum formosum*), a piscivore, abundance was significantly greater. The next size class (2-5 cm) showed a significant difference in the number of fishes recorded among quads with shelter. The breakdown of size classes into 0-2 cm and 2-5 cm and resulting data indicates that young recruits who have survived the first few days or weeks may become better able to utilize available microhabitats for shelter from predation.

Fishes greater than 30 cm were the least abundant of all the size classes across the four treatments. Although 155 fishes in this largest size class were counted on the Empty treatment quads, amber jacks (*Seriola dumerili*) and blue runners (*Caranx crysos*) accounted for 108 of the fish; the next largest total of carangids in this size class was on the Large treatment with only 28 fish counted. As the Empty treatment quads did not have cage material or block functioning as internal refuge, and the totals were recorded over multiple censuses, these transient predators may have been targeting low shelter quads with more accessible prey resulting in an increased foraging efficiency.

The next most abundant fishes in the 30^{+} cm size class across all four treatments were parrotfish (Scaridae) and other wrasses (Labridae). In general, the lack of significant differences in the abundance and species richness for fishes in the size class 30^{+} cm can be attributed to fishes that may have been using the internal refuge of Small, Mixed, and Large treatment quads for shelter as juveniles until reaching a size where they were less prone to predation, which allowed them to move about the sand flat to forage (hogfish, grunts, and snappers), move between quads if disturbed by the presence of divers (parrotfish), or emigrate to the natural reef areas to the east or west.

Of the remaining top ten species that contributed most to dissimilarity in the fish assemblage (Table 2), cottonwicks (*Haemulon melanurum*) were in significantly greater numbers on the Large treatment quads than the other three treatments and significantly more on the Mixed than Empty (Table 1). These fish were in the 2-5 and 5-10 cm size classes so were perhaps finding more refuge on the modules with large fill (cinder block) than small fill (plastic cage). Hixon and Beets (1989, 1993) showed the size of a fish using a reef for shelter was relative to the size of available refuge. However, here it appears that larger refuge (cinder blocks) was almost always preferable for abundance, species richness, and biomass in each size class. A refuge difference between this study and Hixon and Beets (1989) was the addition of shelter around the cinder blocks (between block and the wall of the module). This extra layer may have acted as a partial predator exclusion device allowing smaller fish to use the enclosed microhabitat as refuge.

Lastly, the abundance of spotted goatfish (*Pseudupeneus maculatus*) was significantly less on the Empty treatment quads compared to the other three treatments which did not differ in abundance. Spotted goatfish are diurnal benthic foragers and so would not be expected to benefit from the increased shelter found in the Small, Mixed, and Large quads. Fecal material from the fish assemblage may be acting as trophic fertilizer (Meyer and Schultz 1985a) for zoobenthos resulting in a greater or more diverse infaunal assemblage (concomitant with the large fish assemblage) upon which the goatfish feed (Pauly et al. 2009). This is counter-intuitive however as invertebrate density has been shown to decrease the closer to artificial reefs one gets (Lindberg 1996; Bortone et al. 1998; Bortone 1999). Further investigation is needed to determine what ecological processes may be creating this difference in goatfish abundance among the various treatments.

Benthic structure with holes, overhangs, and shadows have been shown to be preferable habitat over less complex reef areas for coral reef fishes (Roberts and Ormond 1987; Hixon and Beets 1989, 1993; Holbrook et al. 2002). Block used in the Large treatment quads created the most habitat with these characteristics within the artificial reef modules.

Although some differences were not statistically significant, in general the Large fill treatment had a greater abundance, species richness, and biomass than the other three treatments. The Small fill treatment provided refuge from predators, in a way similar to Gilliam's (1999) study, but the quads lacked any other type of solid internal structure. The Mixed fill treatment quads had one module with block that did create holes and this treatment was more often similar in fish assemblage structure to the Large fill than the other two treatments.

Although 9 – 12 species of fishes were found to be unique to a specific treatment in this study, none of the species were consistently abundant on any specific treatment such that structural differences in the treatment complexities should be attributed to the species settlement. Thus, occurrences of unique species, with the exception of transient predators (e.g. rainbow runner, *Elagatis bipinnulata*) is likely attributable to stochastic processes, i.e. settlement, mortality, or illness.

The comparison of fish assemblages among treatments confirms previous studies (Hixon and Beets 1989; Eklund 1996; Sherman et al. 2001) that greater fish abundance and/or species richness is associated with an increase in complexity. Thus, internal complexity of filled treatments (Small, Mixed, and Large) was associated with a greater number of resident fishes that may require refuge (e.g. juvenile grunts, bluehead wrasse, slippery dicks).

The significant differences in fish assemblages created by the four quad complexities supports the H_1: "fish assemblages associated with ARs result from a difference in the artificial reef structural complexity". Although some studies indicate that fish recruitment and subsequent assemblage structures may be influenced to a greater extent by stochastic recruitment (Sale et al. 1984) and post-settlement movement (Caley et al. 1996), this study supports other findings that associate larger reefs with shelter from predators with a more diverse fish assemblage (Luckhurst and Luckhurst 1978; Roberts and Ormond 1987; Caselle and Warner 1996) than smaller, less complex reefs.

4.2 Coral recruitment

Although Banks et al. (2008) reported almost 40 species of scleractinian corals found along the southeast Florida reef tracts, individual studies have only reported 27 – 30 species (Goldberg 1973, Moyer et al. 2003). My study was done between the inner and middle reef tracts in Broward County where 24 coral species have been identified with an average of 8.8 species per monitoring station (Gilliam 2007). Off Palm Beach County, immediately north of Broward County, Goldberg (1973) recorded 15 species of coral along the middle reef tract; *Oculina diffusa, Solenastea hyades, Dichocoenia stokesii*, and *Montastrea cavernosa* were the most common. *Porites* and *Agaricia* contributed only about 5.6% of coral cover.

Vermeij (2005) deployed settlement plates in Curacao on reefs with an estimated 20 – 30% coral cover and, after three years, found only 80 recruits from five species, four of which were brooders. In contrast, scleractinian coral cover on the reef tracts off Broward County is low, typically < 6%, and *Montastrea cavernosa* generally dominates as the major hermatypic scleractinian (Moyer et al. 2003). However, mean coral cover has been determined to be as low as 0.4% for the middle reef and 0.3% for the outer reef at specific sites with *M. cavernosa* and *Porites astreoides* contributing most to the coral cover at these locations (Gilliam 2007).

Pallet ball modules are reported to have a surface area of 7 m^2 (Barber 2007). Eighty modules, the number surveyed for coral recruitment, would give an approximate total surface area of 560 m^2 resulting in a recruitment density of 0.3 recruits per m^2 of substrate. In contrast, ATT/DERM artificial reef modules, 30 meters north west of the study site, had a reported density of 9.6 recruits per m^2 (Deis and Kosmynin in press). Although coral recruitment is highly variable spatially and temporally, some differences should be noted between the two studies. First, ATT/DERM modules were in the water for 4.5 years before final assessment, compared to just over 3 years for this study. Second, nine ATT/DERM modules were used as a nursery for displaced corals, had 193 coral transplants and, although these nursery modules were not monitored in the Deis and Kosmynin sutdy, resulted in a density on the modules much greater than the adjacent hard-bottom areas. *Dichocoenia stokesii*, *Colpophyllia natans*, *Diploria labyrinthiformes*, *Diploria strigosa*, *Meandrina meandrites*, and *Eusmillia fastigiata* were not found on the adjacent hard-bottom area, but were present as both transplants and recruits on the modules. Finally ATT/DERM modules were placed much closer to the natural reef tract (approximately 5 m) (author, personal observation) than this study's modules (30 m) (Figure 4). Reyes and Yap (2001) found that recruitment to settlement plates on natural reef was significantly greater than plates placed less than 5 m from the substrate.

Bare substrate along the southeast Florida reef tracts is estimated to cover between 50% and 70% (Banks et al. 2008), so overgrowth or competition for suitable substrate on the quads was not expected to be a concern in this study. While not quantified in this study, however, macro-algae and encrusting sponges became prevalent on all the modules over the course of the study, thus reducing the amount of available space for coral recruitment (author, personal observation).

The brooding corals *Agaricia* and *Porites* made up the greatest number of recruits in this study. They have also been found to be the dominant genera of recruits in the Florida Keys (Chiappone and Sullivan 1996; Edmunds et al. 2004; Moulding 2007), as well as in recruitment studies conducted around the Caribbean (Bak and Engel 1979; Rogers et al. 1984; Smith 1992; Edmunds 2000; Vermeij 2005).

At the time the study's coral recruit assessment was conducted, Gilliam et al. (2004) found *Sideratrea siderea*, *S. radians*, *Montastrea cavernosa*, *Porites astreoides*, *Stephanocoenia intersepta*, and *Millipora alcicornis* to be the most numerous species in Broward, with *Agaricia agaricites* as the 13th most abundant. These findings differ from the coral assemblage found during this study, but it has been shown that recruitment patterns of juvenile scleractinian corals often do not reflect the adult coral community (Bak and Engel 1979, Edmunds 2000). *Agaricia* and *Porites* are hermaphroditic and self-fertilizing which is a reproductive strategy that allows for multiple recruitment opportunities on new substrate (Szmant 1986). Massive corals (e.g. *Meandrina meandrites*, *Montastrea* spp., *Siderastrea* spp.) typically reproduce only once annually and thus have limited recruitment opportunities per year. The scleractinian coral *Siderastrea* spp. was the second most common benthic colonizer and most common hermatypic coral on artificial reef modules off Miami-Dade County, but this was attributed to a large recruitment pulse during the fourth and fifth year of that study (Thanner et al. 2006). The lack of recruits from these corals may indicate that a longer time horizon (> 5 yrs) is required before a natural benthic community structure similar to the surrounding hard-bottom develops.

Studies report a wide range (12-36 mm yr^{-1}) of extension rates for juvenile scleractinian corals (Bak and Engel 1979; van Moorsel 1988; Chiappone and Sullivan 1996). Gomez et al. (1982) determined rates as high as 45 mm yr^{-1}. Moulding (2006) assumed that juveniles < 10 mm had recruited within the last year while Bak and Engel (1979) suggested this size was 1-3 yrs old. Some authors include partial mortality with extension rates in determining average annual growth of the colony (van Moorsel 1988) resulting in rates as low as 2 mm yr^{-1} (Edmunds 2000). Additionally, authors may not differentiate extension rates by species or genera (Gomez et al. 1982, Edmunds 2000). Van Moorsel (1988) assumed equal extension rates (18-28.8 mm yr^{-1}) for all scleractinian species as recruits initially expand 2-dimensionally over the substratum and this early expansion doesn't allow for much inter-specific variation. Also, some authors assume maximum extension rates are only expected under conditions of low or no stress to the coral recruits (van Moorsel 1988), however rates have been shown to actually increase from oligotrophic to mesotrophic conditions (Edinger et al. 2000) which are generally considered less optimal for coral growth. The majority (96%) of the coral recruits recorded in this study are ≤ 40 mm in diameter. Based on an average, conservative, diameter-extension rate of 12 mm yr^{-1}, recruits in the size range of 39-51 mm would have settled during the first year after deployment, recruits in the size range of 26-38 mm would have settled in the 2nd year, recruits 13-25 mm would have settled in the 3rd year, and recruits smaller than 13 mm would have settled within the last few months before being counted. However, this extension rate would not be a reliable estimate for coral recruits in the larger size ranges (39-51 mm) as estimated time from settlement exceeds the time the modules have been deployed. With modules in the water for 3.5 years (November 2000 – May 2004), the largest recruit, *Porites* with a 66 mm diameter, would give a conservative growth rate of approximately 19 mm yr^{-1}. However, it has been theorized some of the “faster” growing recruits are in fact multiple larvae that settled close together and fused (Harrison and Wallace 1990) and *Porites porites* planula have been shown to settle close to other *P. porites* planula and fuse into a single larger “recruit” (Lewis 1974).

Another factor possibly affecting coral recruitment is the accumulation of a microbial biofilm on substrates which may promote settlement of marine invertebrate larvae (Wieczorek and Todd 1989; Anderson 1996). The biofilm, which consists of bacteria, algae, diatoms and other microbial biota, may take a year or more to develop (Anderson 1996). This may have reduced the amount of time modules used here were conducive to coral recruitment from 3.5 to 2.5 years. If the largest recruit (*Porites*, 66 mm) settled at the end of the first year after deployment (Oct. – Nov. 2001), it would have had a conservative growth rate estimate of 26.4 mm yr^{-1}. Using this growth rate, approximately 84% of the recruits (156, ≤30 mm) would have settled within the last year of the study. Estimates of growth may need to be calculated on a site specific basis and specifically on corals that have recruited to natural or artificial substrate.

A number of fish families, including damselfishes (Pomocentridae), parrotfishes (Scaridae), and surgeonfishes (Acanthuridae), have been known to kill juvenile corals either intentionally or through incidental browsing and grazing (Randall 1974; Goreau et al. 1981; Hixon 1983; Harriott 1985). Parrotfishes have been reported to cause mortality on > 13% of the *Porites astreoides* colonies in Belize although the areas grazed may have been selective and coral mortality incidental as the fishes appeared to be targeting coral areas with higher densities of macroboring organisms (Rotjan and Lewis 2005). Thus, the sharp decrease in number of coral recruits from 35 to 8 in the size classes 26-38 mm to 39-51 mm, respectively, could be a factor of post-recruitment mortality (although Birkland (1997) found that Caribbean fishes may intentionally avoid corals greater than 20 mm in diameter). While there was no direct observation in this study of parrotfishes grazing on juvenile corals, there was evidence of parrotfish foraging on the quads in the form of apparent parrotfish bite marks on the concrete used to secure the settlement plates to the modules.

Cyanobacteria and macroalgae can inhibit coral recruitment at a level that may allow abundant macrophytes to perpetuate a phase shift in the local ecosystem (Kuffner et al. 2006). An extensive *Lyngbya* spp. bloom, likely containing a previously unknown cyanobacteria species, occurred on the reef tracts beginning in 2002 and continued for three years (Paul et al. 2005). The artificial reef modules in this study became heavily overgrown with cyanobacteria during the summer of 2003 and the growth persisted until the spring of 2004. During this time period, all of the modules were so heavily overgrown with *Lyngbya* spp. that settlement plates were not visibly distinguishable from the actual modules. *Lyngbya majuscula* has been shown to negatively influence coral larval settlement through allelopathy and possibly physical interactions such as entanglement in hair-like filaments of the algal tufts (Kuffner and Paul 2004). Additionally, *Lyngbya* spp. has been shown to specifically inhibit settlement and post-settlement survival of *Porites astreoides* (Kuffner et al. 2006; Paul et al. 2008) and significantly affect percent stony coral cover (Semon et al. 2008).

Crustose coralline algae (CCA) has been shown to positively influence coral settlement (Morse et al. 1988; Morse and Morse 1991; Morse et al. 1994; Morse and Morse 1996), but Kuffner and Paul (2004) found *Lyngbya majuscula* tufts concealed dead crustose coralline algae in an anoxic environment. Thus cyanobacteria blooms may create conditions less favorable for coral settlement for an extended period after the blooms have retreated and until CCA recolonizes the substrate.

The total number of recruits counted on the quads was an underestimate as interior surfaces of the modules and concrete block were not surveyed and at least one scleractinian coral recruit (*Diploria* sp.) was observed on a large-refuge block. Coral larvae are known to actively select microhabitats (Edmunds et al. 2004) and studies have shown settlement preferences on cryptic substrates (Carleton and Sammarco 1987; Harrison and Wallace 1990) although the tendency for settlement on the upper surface of the substrate may increase with depth due to a reduction in light intensity and possible competition with macroalgae (Edmunds et al. 2004).

4.3 Coral recruitment and fish assemblages

Relatively few families of fishes (Ephippidae, Chaetodontidae, Pomacentridae, Labridae, Scaridae, Bleniidae, Acanthuridae, Balistidae, Diodontidae, Tetraodontidae) are known to eat coral polyps or otherwise have a direct effect on coral colonies (Randall 1974; Patton 1976). Although considered a major reef-fish corallivore, Pacific butterflyfishes (Chaetodontidae) actually consume a negligible portion (Bouchon-Navaro and Harmelin-Vivien 1981) of coral. Triggerfishes (Balistidae) and puffers (Tetraodontidae) are the only other larger fishes known to regularly consume corals. Of these families, only Chaetodontidae and Pomacentridae showed a correlation with coral recruitment in this study.

Examining reef fish diversity and benthic coverage in the Red Sea, Roberts and Ormand (1987) found, out of five fish families observed, only one (chaetodontids) correlated with live coral cover. In this study, all four butterflyfish species combined (426 total) showed a significant correlation with corals assumed to have recruited in the first year and corals assumed to have recruited in the last year of the study. even though the total number of reef butterflyfish (*Chaetodon sedentarius*) counted during the study was 365, the lowest count of fishes among the selected-species of potential corallivores (butterflyfishes, damselfishes, grunts, parrotfishes, pufferfishes, surgeonfishes, triggerfishes, and wrasses) (see Results 3.2.2). Additionally, the reef butterflyfish, which was the only chaetodontid in the top 90% of species by abundance, showed the same correlation with first year and last year coral recruits. As the number of reef butterflyfish counted was almost six times the total number of other chaetodontids combined (365 of 426, Table 1), the correlation of all butterflyfish species combined with coral recruits is probably a result of the high proportion of reef butterflyfish in the analysis. Hourigan (1988) found that corallivorous butterflyfish in the Pacific prefer to feed on the same coral species, but coral feeding preference, if any, of the reef butterflyfish is lacking. Butterflyfish (*Chaetodon* sp.), however, have been observed in other areas off Broward County feeding on recently transplanted *Solanastea bournoni* colonies with grazed polyps being distinguishable from ungrazed polyps (D. Gilliam personal communication).

Most *Stegastes* spp. (Pomacentridae) are known to be aggressive when defending their territory and territorial damselfish can have an effect on coral communities either by removing polyps or killing corals when establishing and defending algal mats that can be up to 1 m in diameter (Hixon 1983). In this study, damselfish showed a significant correlation with the corals (39-51 mm) assumed to have recruited during the first year after deployment. Although bicolor damselfish (*Stegastes partitus*) were the most abundant of the damselfishes by far (402 of 595 total pomacentrids, Table 1), six species of *Stegastes* were counted on the quads.

Grunts showed a correlation with juvenile corals assumed to have recruited in the third year (13-25 mm size class) after deployment with the total of all haemulids combined (selected-species) and total haemulid species in the top 90% of fish by abundance. Six species/taxa of grunts were in the top 90% category: cottonwick (*Haemulon melanurum*), french (*Haemulon flavolineatum*), tomtate (*Haemulon aurolineatum*), white (*Haemulon plumierii*), porkfish (*Anisotremus virginicus*), and juvenile grunts (*Haemulon* spp.). Grunts are diurnal planktivores as juveniles and feed while hovering over the reef. Additionally, adult grunts are benthic nocturnal feeders (Pauly et al. 2009) and are not known to prey on corals, but during the day they hover over the reefs and thus can influence the growth of corals through their excrement (Meyer et al. 1983; Meyer and Schultz 1985a, b). Meyer and Shultz (1985a) found that daily excretion and defecation by grunts doubled the amount of NH_4^+, a form of nitrogen readily usable by coral zooxanthellae (Muscatine and D'Elia 1978), and the time of maximum coral growth for the scleractinian coral *Porites furcata* occurred during the time of maximum input from the grunt population.

Both the selected-species total abundance and the selected-species' top 90% abundance of fishes showed a correlation with coral recruits assumed to have recruited in the third year after deployment, but as grunts made up 64% of the total abundance and 66% of the top 90% abundance, it is probable this correlation is an artifact of the grunts' dominant presence in these categories. Additionally, no other fish species were correlated with coral recruits in the 13-25 mm size class.

Although some correlations were statistically significant, growth rates of corals did not appear to be correlated with the fish assemblages as the seven largest corals (> 40 mm) were found on three Small and four Large treatment quads, which were significantly different in terms of fish species richness and biomass. The Small and Large, along with the Mixed treatment, were not significantly different in terms of fish abundance, but no corals in the larger size classes were found on Mixed treatment quads.

While few direct correlations were determined between fish assemblages and coral recruits, there were intriguing parallels between the two. The Empty treatment quads had fewer fish in terms of abundance, species richness, and biomass and similarly, these same quads had fewer coral recruits than the other three treatments. Species richness for coral recruits was not quite so distinct, as almost 25% were identified only to the genus level, but the Large treatment quads, which had more fish abundance and greater biomass, were significantly more specious in coral recruits than the Empty, Small, or Mixed treatments. Chabanet (1997) theorized that the relationship between fish abundance/species richness and corals is more a factor of increased microhabitats than diversity or abundance of corals and Cabaitan et al. (2008) showed an increase in coral cover, which also increased complexity, resulted in an increase in fish abundance and species richness. Roberts and Ormond (1987), however, found benthic biological diversity, but not live coral cover, was more highly correlated with fish species richness than structural complexity in the Red Sea. Refuge, in the form of multiple sized holes, accounted for much of the variance in fish abundance. Later studies (Holbrook et al. 2006; Feary et al. 2007; Holbrook et al. 2008) in the Indo-Pacific have found that fish assemblages were strongly influenced by changes in the amount of live coral cover, rather than structural complexity, when percent coral coverage was less than 10%. The influence on fish species richness and abundance of percent live coral above this threshold was much more difficult to detect, indicating other ecological processes may take over the influence of these fish assemblages.

Although there were some correlations between certain fish species (grunts, damsels, and butterflyfish) and juvenile corals, due to the low number of recruits it is unclear whether the correlations are real or apparent statistical significance was a chance artifact of the data. Even with data transformation, the data did not meet all of the assumptions of a parametric analysis, thus caution should be used in interpreting the significance of these results. Non-parametric analyses (Spearman-Rank correlation) were completed on what must be considered a small data set and the statistical results may not correctly reflect relationships between fish abundance and coral recruitment.

Additional coral recruit data may be needed to elaborate on the results presented here so as to positively support or not support H_2: ”different fish assemblages affect the recruitment of coral onto artificial reefs”. The data required could be in the form of greater recruit abundance and/or a longer monitoring schedule to possibly allow a more diverse coral community to develop, one that is more similar to the surrounding reef tracts.

4.4 Coral attractants

Due to the low number of coral recruits on settlement plates, rigorous statistical analysis could not be performed. Similar difficulties in statistical testing for patterns of density, success of recruits, and distinguishing genus or species level effects arise whenever there is a paucity of juvenile corals (Edmunds et al. 2004). However, there are noteworthy trends relating to $CaCO_3$ and transplant treatments.

Settlement plates treated with $CaCO_3$ accounted for 37% of the total recruits found on all plates. This was almost twice the number of recruits found associated with iron and transplant treatments and approximately 60% more than found on controls. Reyes and Yap (2001) used a similar approach in the Philippines of pressing $CaCO_3$ sand onto the surface of settlement plates in addition to mixing the sand with the concrete, but did not did not find any difference between $CaCO_3$ treated plates and standard concrete controls. Coral cover at their study site was estimated at 40 – 60%, so presumably there would be a concomitant increase in the supply of coral larvae available for settlement compared to the location in this study.

Plates neighboring coral transplants showed a greater and more complex trend. As the plate itself did not contain the actual transplant, the influence of live coral cover may have a much broader effect. All 40 *Montastrea cavernosa* transplants survived the length of the study with no tissue die-off and growth ranging from 7 – 219% of the original transplant size. However, 29 (73%) of *Meandrina meandrites* transplants suffered some degree of mortality. Fifteen (38%) transplants experienced tissue loss (partial mortality) of 20 – 95% while an additional 14 transplants (35%) suffered total mortality. Colonies that suffered mortality were distributed amongst the treatment types: Empty – 3 partial mortality and 3 total mortality; Small – 6 partial and 3 total; Mixed – 1 partial and 5 total; Large – 5 partial and 3 total. Even with the reduction in the number of transplants due to partial or total mortality, of the 186 total coral recruits in the study, transplant modules (M1) contained 102 (55%) of the recruits compared to 68 (37%) recruits found on the other modules (M3) sampled. The remaining 16 (8%) recruits were counted on settlement plates attached to M2 or M4 of each quad. Although all 40 of the transplant attractant modules (M1) were sampled, the M3 sample contained a mixture of the remaining attractant treatments: $CaCO_3$ –

14 modules, iron – 15 modules, control – 11 modules. If it is assumed that these three treatments had little effect influencing coral recruitment over the entire module, and thus all considered controls, then the presence of a single coral transplant enhanced recruitment onto the M1 modules. Lewis (1974) examined three corals (*Agaricia agaricites, Porites astreoides*, *Favia fragum*) for intra-specific influence of recruits to settle close together or near a larger colony. Although he determined that clumping arrangements of *A. agaricites* and *P. astreoides* colonies were due instead to splitting of larger colonies, it was concluded that *F. fragum* clumps were the result of attraction and settlement of larvae near already established colonies. Unlike *P. astreoides*, Goreau et al. (1981) found that patchiness in total settled populations of *Porites porites* was the result of non-random settling and individuals were more likely to attach near other individuals than away from them. Whether this was a result of direct interaction between the recruits or indirect tactile and/or chemosensory tropisms remained unclear.

Although non-parametric statistical analysis (Kolmogorov-Smirnov two sample test) did not reveal a significant difference between the sampled modules, transplantation of corals onto artificial reefs has been suggested as a means to stimulate coral growth (Oren and Benayahu 1997). Clark and Edwards (1994) recommend transplantation only take place where natural coral recruitment is unlikely to result in restoration, but as the addition of a single coral may contribute to the restoration success, further investigation into the effects of transplantation may be warranted.

Due to a low number of coral recruits on the attractant substrates preventing rigorous statistical analysis, there appears to be insufficient data to support or not support H_3: ”coral recruitment to settlement plates can be influenced by substrates or attractants”. Redesign of the methodology in an attempt to allow greater coral recruitment (e.g. increased sampling period to allow more coral to recruit, placement of the substrate attractants to areas of greater coral recruitment) is needed to gather more data as conclusive evidence.

5. Conclusion

The purpose of my study was to examine possible methods for enhancing coral reef restoration by testing several hypotheses. The summary of this work and the resulting recommendations are provided below.

Sherman et al. (2002) found internal complexity in artificial reefs creates a more diverse fish assemblage than less complex reefs with a simple void space. This study supports those findings and further shows that differing internal structural complexity of artificial reefs can lead to significantly different fish assemblages. Thus, artificial reefs used to create habitat on degraded or damaged reefs should likely incorporate similar design features which mimic shadowed overhang areas found under large coral colonies or reef ledges. Additionally, although some studies indicate that fish recruitment and subsequent assemblage structures may be influenced to a greater extent by stochastic recruitment (Sale et al. 1984) and post-settlement movement (Caley et al. 1996), this study supports other findings that associate larger reefs containing shelter from predators with a more diverse fish assemblage (Luckhurst and Luckhurst 1978; Roberts and Ormond 1987; Caselle and Warner 1996) than smaller, less complex reefs. Thus, the use of multiple artificial reef modules placed close together can create a more specious fish assemblage than smaller more isolated modules.

Due to funding, the artificial reefs in this study were only monitored for 3 years. If a more complete understanding of the artificial reef function is desired, I recommend the consideration of extended monitoring time frames as even large artificial reefs still may not develop assemblages that resemble the natural reefal environment for 5^{+} years (Thanner 2006).

The limited amount of coral recruitment to the modules in this study was dominated by brooding corals (*Porites astreoides* and *Agaricia agaricites*, respectively) which have multiple reproductive cycles per year. A longer (> 5 yrs) or possibly delayed monitoring schedule should be used to establish if a particular artificial reef restoration project is successful relative to developing a natural benthic community structure. Also, placement of the artificial reefs can be critical as studies indicate that substrate placed closer to the natural reef have greater recruitment rates than those placed out in sand habitat (Reyes and Yap 2001; Deis and Kosmynin in press). The placement of artificial reef substrate away from coral larval sources may adversely affect recruitment onto the artificial reefs as coral larvae are essentially passive drifters in the water column (Carlon and Olson 1993; Largier 2004; Ritson-Williams et al. 2008). With the predominant current flow in the study area from south to north (Soloviev et al. 2001), the chance of planulae being carried from the reefs on the east and west sides of the artificial reef array were reduced.

Studies have shown that fishes can have a positive effect on coral growth (Meyer and Schultz 1985a, b) and likewise, coral can positively affect fish assemblage structure (Roberts and Ormond 1987; Hixon 1997; Holbrook et al. 2006; Holbrook et al. 2008). In this study, damselfishes, grunts, and butterflyfishes showed a correlation with coral recruits. However, due to the artificial reef structure and possibly the cryptic nature of initial coral settlement (Carleton and Sammarco 1987; Harrison and Wallace 1990), the number of coral recruits counted is an underestimate as it was not possible to survey the inside of the quads nor the concrete block for recruits. The correlation of coral recruitment with fishes may include more fish families, but a different structural design that would allow census of all coral recruits would be required.

Data on the use of attractants to enhance coral recruitment, while limited in recruitment numbers, did yield results worth further investigation. Plates treated with $CaCO_3$ had almost twice the number of recruits than the iron or transplant treatments, however the overall lack of coral recruitment may have been the result of one or more multiple factors. The amount of attractant used on the plates was a "best guess" and may have lacked the quantity required to be effective over multiple recruitment seasons. After a relatively short time in the water, non-coral benthic organisms may have overgrown the plates sufficiently to prevent the attractant (either $CaCO_3$ or iron) from being detected by coral planula. Also, placement of the artificial reefs in an area of open sand habitat may have resulted in fewer planula being delivered to the study area. Concrete artificial reefs made with large limestone, or possibly iron, aggregate and placed close to the natural reef should be used as a next step in determining the effectiveness of these coral attractants.

Although over 70% of the *Meandrina meandrites* coral transplants suffered partial or total mortality over the length of the study all of the *Montastrea cavernosa* transplants survived. As 55% of the coral recruits counted were found on the transplant modules, even the presence of one coral may have a positive effect on coral recruitment. Also, low levels of coral cover (5-10%) have been shown to have a significant effect on fish abundance and species richness (Holbrook et al. 2008). Although transplantation is a time-consuming and expensive effort (Edwards and Clark 1999), it appears the addition of relatively few massive slow-growing corals could be highly beneficial to a restoration project in terms of enhancing coral recruitment and the fish assemblage.

A summary of restoration lessons learned in this study are as follows:

1) Artificial reefs should be designed with fish microhabitat consisting of holes, overhangs, and shadowed refuge. The absence of refuge in Empty treatment quads resulted in significantly less fish species richness, abundance, and biomass. Although not always significantly greater, Large fill treatments with holes and shadowed overhangs were generally higher in these fish assemblage measures.

2) Artificial reefs intended to develop a community structure similar to nearby reefs should be placed near the natural hard-bottom or reef that functions as a source of coral larvae. Such placement in close proximity to natural reef areas would increase the probability of coral settlement onto the artificial reefs. The artificial reefs used in this study had fewer coral recruits than other artificial reefs in the area or nearby natural hard-bottom.

3) Create artificial reefs using large limestone aggregate. The number of coral recruits was greater on the $CaCO_3$ (limestone) treated settlement plates than the other three treatments, but the amount of $CaCO_3$ used was small and easily overgrown by macroalgae or other benthic settlers, thus possibly reducing any attractant effectiveness.

4) Finally, although typically an expensive and labor intensive restoration technique, selective transplantation of massive corals onto artificial reefs would be beneficial to further enhance coral settlement and possibly the development of a more diverse fish assemblage structure. Although significant mortality occurred with coral transplants used in this study, each transplant modules had at least one transplant and more coral recruits were found on these modules than the other modules censused. However, as species specific mortality may occur using a coring methodology, species selected should be researched before coring or alternative methods of collecting suitable corals for transplantation should be considered.

6. Literature

Abelson A, Shlesinger Y (2002) Comparison of the development of coral and fish communities on rock-aggregated artificial reefs in Eilat, Red Sea. ICES J Mar Sci 59:S122-S126

Alevizon WS, Gorham JC (1989) Effects of artificial reef development on nearby resident fishes. Bull Mar Sci 44:646-661

Ambrose RF, Swarbrick SL (1989) Comparison of fish assemblages on artificial and natural reefs off the coast of southern California. Bull Mar Sci 44:718-733

Anderson MJ (1996) A chemical cue induces settlement of Sydney Rock oysters, *Saccostrea commercialis*, in the laboratory and in the field. Biol Bull 190:350-358

Arena PT (2005) The structure of fish assemblages on vessel-reefs and adjacent natural reefs in southeast Florida. Ph.D. Dissertation, Nova Southeastern University, p155

Atrigenio MO, Alino PM (1995) Abundance & distribution of juvenile coral recruits & their implications on community structure. Philippine Scientist

Bak RPM, Engel MS (1979) Distribution, abundance and survival of juvenile hermatypic corals (Scleractinia) and the importance of life history strategies in the parent coral colony. Mar Biol 54:341-352

Banks KW, Riegl BM, Shinn EA, Piller WE, Dodge RE (2007) Geomorphology of the Southeast Florida continental reef tract (Miami-Dade, Broward, and Palm Beach Counties, USA). Coral Reefs 26:617-633

Banks KW, Riegl BM, Richards VP, Walker BK, Helmle KP, Jordan LKB, Phelps J, Shivji MS, Spieler RE, Dodge RE (2008) The reef tract of continental southeast Florida (Miami-Dade, Broward and Palm Beach Counties, USA). In: Riegl BM, Dodge RE (eds) Coral Reefs of the USA. Springer, pp175-220

Barber T (2007) Reef Ball Foundation Inc. Technical Specifications For Reef Balls

Baynes TW, Szmant AM (1989) Effects of current on the sessile benthic community structure of an artificial reef. Bull Mar Sci 44:545-566

Beets J (1989) Experimental evaluation of fish recruitment to combinations of fish aggregating devices and benthic artificial reefs. Bull Mar Sci 44:973-983

Bell JD, Galzin R (1984) Influence of live coral cover on coral-reef fish communities. Mar Ecol Prog Ser 15:265-274

Birkeland C (1997) Geographic differences in ecological processes on coral reefs. In: Birkeland C (ed) Life and Death of Coral Reefs. Chapman and Hall, New York, pp273-287

Bohnsack JA (1983) Species turnover and the order versus chaos controversy concerning reef fish community structure. Coral Reefs 1:223-228

Bohnsack JA, Talbot FH (1980) Species-packing by reef fishes on Australian and Caribbean reefs: an experimental approach. Bull Mar Sci 30:710-723

Bohnsack JA, Harper DE (1988) Length-weight relationships of selected marine reef fishes from the Southeastern United States and the Caribbean. NOAA Tech Memo NMFS-SEFC-215:1-31.

Bohnsack JA, Harper DE, McClellan DB, Hulsbeck M (1994) Effects of reef size on colonization and assemblage structure of fishes at artificial reefs off southeastern Florida, U.S.A. Bull Mar Sci 55:796-823

Bortone SA (1999) The impact of artificial reef fish assemblages on their potential forage area: lessons in artificial reef study design. Proceedings of the Florida Artificial Reef Summit:82-85

Bortone SA, Martin T, Bundrick CM (1994) Factors affecting fish assemblage development on a modular artificial reef in a northern Gulf of Mexico estuary. Bull Mar Sci 55:319-332

Bortone SA, Cody RP, Turpin RK, Bundrick CM (1998) The impact of artificial-reef assemblages on their potential forage area. Ital J Zool 65:265-267

Bouchon-Navaro Y, Harmelin-Vivien ML (1981) Quantitative distribution of herbivorous reef fishes in the Gulf of Aqaba (Red Sea). Mar Biol 63:79-86

Brickhill MJ, Lee SY, Connolly RM (2005) Fishes associated with artificial reefs: attributing changes to attraction or production using novel approaches. J Fish Bio 67 (Supp. B):53-71

Brock RE, Kam AKH (1994) Focusing the recruitment of juvenile fishes on coral reefs. Bull Mar Sci 53:623-630

Brownlee AS, Kosmynin VN, Moulding AL, Banks KW, Fisher LE, Gilliam DS (2008) Fish predation on juvenile stony coral, *Siderastrea siderea*, transplanted offshore Southeast Florida, USA. 11th ICRS:528

Butler IV MJ, Hunt JH, Herrnkind WF, Childress MJ, Bertelsen R, Sharp W, Matthews T, Field JM, Marshall HG (1995) Cascading disturbances in Florida Bay, USA: cyanobacteria blooms, sponge mortality, and implications for juvenile spiny lobsters *Panulirus argus*. Mar Ecol Prog Ser 129:119-125

Cabaitan PC, Gomez ED, Alino PM (2008) Effects of coral transplantation and giant clam restocking on the structure of fish communities on degraded patch reefs. J Exp Mar Bio Eco 357:85-98

Cairns J, J (1991) The status of the theoretical and applied science of restoration ecology. The Envir Professional 13:186-194

Caley MJ, Carr MH, Hixon MA, Hughes TP, Jones GP, Menge BA (1996) Recruitment and the local dynamics of open marine populations. Annu Rev Ecol Syst 27:477-500

Carleton JH, Sammarco PW (1987) Effects of substratum irregularity on success of coral settlement: quantification by comparative geomorphological techniques. Bull Mar Sci 40:85-98

Carlon DB, Olson RR (1993) Larval dispersal distance as an explanation for adult spatial pattern in two Caribbean reef corals. J Exp Mar Biol Ecol 173:247-263

Carr MH, Hixon MA (1997) Artificial reefs: the importance of comparisons with natural reefs. Fisheries 22:28-33

Caselle JE, Warner RR (1996) Variability in recruitment of coral reef fishes: the importance of habitat at two spatial scales. Eco 77:2488-2504

Chabanet P, Ralambondrainy H, Amanieu M, Faure G, Galzin R (1997) Relationships between coral reef substrata and fish. Coral Reefs 16:93-102

Chandler CR, R. M. Sanders J, A.M. Landry J (1985) Effects of three substrate variables on two artificial reef fish communities. Bull Mar Sci 37:129-142

Chiappone M, Sullivan KM (1996) Distribution, abundance and species composition of juvenile scleractinian corals in the Florida reef tract. Bull Mar Sci 58:555-569

Clark S, Edwards AJ (1994) Use of artificial reef structures to rehabilitate reef flats degraded by coral mining in the Maldives. Bull Mar Sci 55:714-744

Clark S, Edwards AJ (1999) An evaluation of artificial reef structures as tools for marine habitat rehabilitation in the Maldives. Aquat Conser: Mar Fresh Eco 9:5-21

Cummings SL (1994) Colonization of a nearshore artificial reef at Boca Raton (Palm Beach County), Florida. Bull Mar Sci 55:1193-1215

Deis DR, Kosmynin VN (in press) Coral recruitment on artificial reefs off Fort Lauderdale, Florida. In press

DERM PBC Final report - Artificial reef habitat and recreational reef fish abundance 78

Doherty PJ, William DM (1988) The replenishment of coral reef fish populations. Oceanogr Mar Biol Annu Rev 26:487-551

Edinger EN, Limmon GV, Jompa J, Widjatmoko W, Heikoop JM, Risk MJ (2000) Normal coral growth rates on dying reefs: are coral growth rates good indicators of reef health? Mar Poll Bull 40:404-425

Edmunds PJ (2000) Patterns in the distribution of juvenile corals and coral reef community structure in St. John, US Virgin Islands. Mar Ecol Prog Ser 202:113-124

Edmunds PJ, Bruno JF, Carlon DB (2004) Effects of depth and microhabitat on growth and survivorship of juvenile corals in the Florida Keys. Mar Ecol Prog Ser 278:115-124

Edwards AJ, Clark S (1999) Coral transplantation: a useful management tool or misguided meddling? Mar Poll Bull 37:474-487

Edwards AJ, Gomez ED (2007) Reef Restoration Concepts and Guidelines: making sensible management choices in the face of uncertainty. Coral Reef Targeted Research & Capacity Building for Management Programme, St. Lucia, Australia 38

Eklund A (1996) The effects of post-larval settlement predation and resource limitation on reef fish assemblages. Ph.D. Dissertation, University of Miami, p148

Fahy EG (2003) Growth and survivorship of *Meandrina meandrites* and *Montastrea cavernosa* transplants to an artificial reef environment, and the effectiveness of plugging core holes in transplant donor colonies. Nova Southeastern University, p123

Fearon RJ, Cameron AM (1997) Preliminary evidence supporting the ability of hermatypic corals to affect adversely larvae and early settlement stages of hard coral competitors. J Chem Eco 23:7:1997

Feary DA, Almany GR, McCormick MI, Jones GP (2007) Habitat choice, recruitment and the response of coral reef fishes to coral degradation. Oecologia 153:727-737

Ferro F, Jordan LKB, Spieler RE (2005) The marine fishes of Broward County, Florida: final report of 1998-2002 survey results NOAA Tech Memo NMFS-SEFSC-532 73pp

Field JG, Clarke KR, Warwick RM (1982) A practical strategy for analyzing multispecies distribution patterns. Mar Ecol Prog Ser 8:37-52

Fitzhardinge RC, Bailey-Brock JH (1989) Colonization of artificial reef materials by corals and other sessile organisms. Bull Mar Sci 44:567-579

Frazer TK, Lindberg WJ (1994) Refuge spacing similarly affects reef-associated species from three phyla. Bull Mar Sci 55:288-400

Freeman JA (2007) Comparison of fish assemblages between mitigation boulder reef and neighboring natural hardbottom in Broward County, Florida USA. Thesis, Nova Southeastern University, p103

Frid CLJ, Clark S (1999) Restoring aquatic ecosystems: an overview Aquat Conser: Mar Fresh Eco 9:1-4

Gilliam DS (1999) Juvenile reef fish recruitment processes in South Florida: a multifactorial field experiment. Ph.D. Dissertation, Nova Southeastern University, p111

Gilliam DS (2007) Southeast Florida Coral Reef Evaluation and Monitoring Project: 2006 Year 4 Final Report 31

Gilliam DS, Dodge RE, Spieler RE, Jordan LKB, Monty JA (2004) Technical Report EPD 05-02 Marine Biological Monitoring in Broward County, Florida: Year 5 Annual Report 88

Gochfeld DJ (1991) Energetics of a predator-prey interaction: corals and coral-feeding fishes. Pac Sci 45:246-256

Golani D, Diamant A (1999) Fish colonization of an artificial reef in the Gulf of Elat, northern Red Sea. Envir Biol Fishes 54:275-282

Goldberg WM (1973) The ecology of the coral-octocoral communities off the Southeast Florida coast: geomorphology, species composition, and zonation. Bull Mar Sci 23:465-488

Gomez ED, Alcala AC, Alcala LC (1982) Growth of some corals in an artificial reef off Dumaguete, Central Visayas, Philippines Kalikasan. Philipp J Biol 11:148-157

Goreau NI, Goreau TJ, Hayes RL (1981) Settling, survivorship and spatial aggregation in planulae and juveniles of the coral *Porites porites* (Pallas). Bull Mar Sci 31:424-435

Gregg KL (1995) Comparisons of three manufactured artificial reef units in Onslow Bay, North Carolina. N Amer J Fish Mgmt 15:316-324

Grove RS, Yuge JE, Sonu C (1980) Artificial reef technology: a strategy for active impact mitigation. Oceans '83 Conference:951-956

Hackney CT (2000) Restoration of coastal habitats: expectation and reality. Eco Eng 15:165-170

Harriott VJ (1985) Recruitment patterns of scleractinian corals at Lizard Island, Great Barrier Reef. Proc 5th ICRS Tahiti 4:367-372

Harriott VJ, Fisk DA (1987) A comparison of settlement plate types for experiments on the recruitment of scleractinian corals. Mar Ecol Prog Ser 37:201-208

Harrison PL, Wallace CC (1990) Chapter 7: Reproduction, dispersal and recruitment of scleractinian corals. In: Dubinsky Z (ed) Ecosystems of the World 25. Elsevier, Amsterdam, pp133-207

Haughton MO, Aiken KA (1989) Biological notes on artificial reefs in Jamaican waters. Bull Mar Sci 44:1033-1037

Herrnkind WF, Butler IV MJ, Hunt JH (1997) Can artificial habitats that mimic natural structures enhance recruitment of Caribbean Spiny Lobster? Fisheries 22:24-27

Hixon MA (1983) Fish grazing and community structure of reef corals and algae: a synthesis of recent studies. Symposia Series for Undersea Research 1:79-87

Hixon MA (1997) Effects of reef fishes on corals and algae. In: Birkeland C (ed) Life and Death of Coral Reefs. Chapman and Hall, New York, pp230-248

Hixon MA, Beets JP (1989) Shelter characteristics and Caribbean fish assemblages: experiments with artificial reefs. Bull Mar Sci 44:666-680

Hixon MA, Beets JP (1993) Predation, prey refuges, and the structure of coral-reef fish assemblages. Eco Mono 63:77-101

Hixon MA, Jones GP (2005) Competition, predation, and density-dependent mortality in demersal marine fishes. Eco 86:2847-2859

Holbrook SJ, Brooks AJ, Schmitt RJ (2002) Predictability of fish assemblages on coral patch reefs. Mar Freshwater Res 53:181-188

Holbrook SJ, Brooks AJ, Schmitt RJ (2006) Relationships between live coral cover and reef fishes: implications for predicting effects of environmental disturbances. Proc 10th ICRS 1:241-249

Holbrook SJ, Schmitt RJ, Brooks AJ (2008) Resistance and resilience of a coral reef fish community to changes in coral cover. Mar Ecol Prog Ser 371:263-271

Hourigan TF (1988) The behavorial ecology of three species of butterflyfishes (family Chaetogontidae). Ph.D. Dissertation, University of Hawai'i, p513

Hudson JH, Diaz R (1988) Damage survey and restoration of M/V Wellwood grounding site, Molasses Reef, Key Largo National Marine Sanctuary, Florida. Proc 6th ICRS Aust 2:231-236

Hughes TP, Jackson JBC (1980) Do corals lie about their age? Some demographic consequences of partial mortality, fission, and fusion. Sci 209:713-715

Hughes TP, Jackson JBC (1985) Population dynamics and life histories of foliaceous corals. Eco Mono 55:141-166

Hughes TP, Connell JH (1987) Population dynamics based on size or age? A coral reef analysis. Am Nat 129:818-829

Jaap WC (2000) Coral reef restoration. Eco Eng 15:345-364

Jara F, Cespedes R (1994) An experimental evaluation of habitat enhancement on homogeneous marine bottoms in southern Chile. Bull Mar Sci 55:295-307

Johnson CR, Preece AL (1992) Damage, scale and recovery in model coral communities: the importance of system state. Proc 7th ICRS Guam 1:606-615

Jordan LKB, Gilliam DS, Spieler RE (2005) Reef fish assemblage structure affected by small-scale spacing and size variations of artificial patch reefs. J Exp Mar Biol Ecol 326:170-186

Kellison GT, Sedberry GR (1998) The effects of artificial reef verticle profile and hole diameter on fishes off South Carolina. Bull Mar Sci 62:763-780

Kinsey DW (1988) Coral reef system response to some natural and anthropogenic stresses. Galaxea 7:113-128

Kuffner IB, Paul VJ (2004) Effects of the benthic cyanobacterium *Lyngbya majuscula* on laval recruitment of the reef corals *Acropora surculosa* and *Pocillopora damicornis*. Coral Reefs 23:455-458

Kuffner IB, Walters LJ, Becerro MA, Paul VJ, Ritson-Williams R, Beach KS (2006) Inhibition of coral recruitment by macroalgae and cyanobacteria. Mar Ecol Prog Ser 323:107-117

Largier J (2004) The importance of retention zones in the dispersal of larvae. Amer Fish Soc Symp 42:105-122

Lewis JB (1974) Settlement and growth factors influencing the contagious distribution of some Atlantic reef corals. Proc 2nd ICRS Brisbane 2:201-206

Lighty RG, Macintyre IG, Stuckenrath R (1978) Submerged early Holocene barrier reef south-east Florida shelf. Nature 276:59-60

Lindberg WJ (1996) Fundemental design parameters for artificial reefs: interaction of patch reef spacing and size. FDEP Tech Rep:71

Luckhurst BE, Luckhurst K (1978) Diurnal space utilization in coral reef fish communities. Mar Biol 49:325-332

McManus JW, Miclat RI, Palaganas VP (1981) Coral and fish community structure of Sombrero Island, Batangas, Philippines. Proc 4th ICRS Manila 2:271-280

Meyer JL, Schultz ET (1985a) Tissue condition and growth rate of corals associated with schooling fish. Limnol Oceano 30:157-166

Meyer JL, Schultz ET (1985b) Migrating haemulid fishes as a source of nutrients and organic matter on coral reefs. Limnol Oceano 30:146-156

Meyer JL, Schultz ET, Helfman GS (1983) Fish schools: an asset to corals. Sci 220:1047-1048

Miller MW, Hay ME (1998) Effects of fish predation and seaweed competition on the survival and growth of corals. Oecologia 113:231-238.

Miller MW, Barimo J (2001) Assessment of juvenile coral populations at two reef restoration sites in the Florida Keys National Marine Sanctuary: Indicators of success? Bull Mar Sci 69:395-406

Morse ANC, Morse DE (1996) Flypapers for coral and other planktonic larvae. BioScience 46:254-262

Morse DE, Morse ANC (1991) Enzymatic characterization of the morphogen recognized by *Agaricia humilis* (Scleractinian coral) larvae. Biol Bull 181:104-122

Morse DE, Hooker H, Morse ANC, Jensen RA (1988) Control of larval metamorphosis and recruitment in sympatric agariciid corals. J Exp Mar Biol Ecol 116:193-217

Morse DE, Morse ANC, Raimondi PT, Hooker N (1994) Morphogen-based chemical flypaper for *Agaricia humilis* coral larvae. Biol Bull 186:172-181

Moulding AL (2007) Coral recruitment in the Florida Keys: patterns, processes, and applications to reef restoration. Ph.D. Dissertation, University of Miami, p157

Moyer RP, Riegl B, Banks K (2003) Spatial patterns and ecology of benthic communities on a high-latitude South Florida (Broward County, USA) reef system. Coral Reefs 22:447-464

Mullineaux LS, Butman CA (1991) Initial contact, exploration and attachment of barnacle (*Balanus amphitrite*) cyprids settling in flow. Mar Biol 110:93-103

Mumby PJ, Harborne AR, Williams J, Kappel CV, Brumbaugh DR, Micheli F, Holmes KE, Dahlgren CP, Paris CB, Blackwell PG (2007) Trophic cascade facilitates coral recruitment in a marine reserve. Proc Natl Acad Sci USA 104:8362-8367

Muscatine LR, D'Elia CF (1978) The uptake, retention, and release of ammonium by reef corals. Limnol Oceano 23:725-734

Neudecker S (1979) Effects of grazing and browsing fishes on the zonation of corals in Guam. Eco 60:666-672

Ogden JC, Ebersole JP (1981) Scale and community structure on coral reef fishes: a long-term study of a large artificial reef. Mar Ecol Prog Ser 4:97-103

Oren U, Benayahu Y (1997) Transplantation of juvenile corals: a new approach for enhancing colonization of artificial reefs. Mar Biol 127:499-505

Patton WK (1976) Animals associates of living reef corals. In: Jones OA, Endean R (eds) Biology and geology of coral reefs. Univ. Queensland, Dep. Geol., St. Lucia, pp33-36

Paul V, Ross C, Ritson-Williams R, Waters L, Arthur K, Gunasekera S, Meickle T (2008) Ecological consequences of cyanobacterial blooms on coral reefs. Proc 11th ICRS Ft Lauderdale:69

Paul VJ, Thacker RW, Banks K, Golubic S (2005) Benthic cyanobacterial bloom impacts the reefs of South Florida (Broward County, USA). Coral Reefs 24:693-697

Pauly D, Kesney-Reyes K, Luna SM (2009) *Pseudupeneus maculatus* Spotted goatfish www.fishbase.org

Perkol-Finkel S, Benayahu Y (2007) Differential recruitment of benthic communities on neighboring artificial and natural reefs. J Exp Mar Biol Ecol 340:25-39

Perkol-Finkel S, Zilman G, Sella I, Miloh T, Benayahu Y (2008) Floating and fixed artificial habitats: Spatial and temporal patterns of benthic communities in a coral reef environment. Est Coast Shelf Sci 77:491-500

Pickering H, Whitmarsh D, Jensen A (1998) Artificial reefs as a tool to aid rehabilitation of coastal ecosystems: investigating the potential. Mar Poll Bull 37:505-514

Pratt JR (1994) Artificial habitats and ecosystems restoration: managing for the future. Bull Mar Sci 55:268-275

Precht WF, Robbart M (2006) Coral reef restoration: the rehabilitation of an ecosystem under siege. In: Precht WF (ed) Coral Reef Restoration Handbook. CRC Press, Boca Raton, pp1-24

Randall JE (1974) The effect of fishes on coral reefs. Proc 2nd ICRS Brisbane 1:159-166

Relini G, Zamboni N, Tixi F, Torchia G (1994) Patterns of sessile macrobenthos community development on an artificial reef in the Gulf of Genoa (Northwestern Mediterranean). Bull Mar Sci 55:745-771

Reyes MZ, Yap HT (2001) Effect of artificial substratum material and resident adults on coral settlement patterns at Danjugan Island, Philippines. Bull Mar Sci 69:559-566

Riegl B, Luke KE (1998) Ecological parameters of dynamited reefs in the northern Red Sea and the relevance to reef rehabilitation. Mar Poll Bull 37:488-498

Rilov G, Benayahu Y (2002) Rehabilitation of coral reef-fish communities: The importance of artificial-reef relief to recruitment rates. Bull Mar Sci 70(1):185-197

Ritson-Williams R, Paul VJ, Arnold S, Steneck RS, Kuffner IB, Walters LJ, Becerro MA (2008) The good, the bad, and the smelly: positive and negative settlement cues for coral larvae. Proc 11th ICRS Ft Lauderdale:69

Roberts CM, Ormond RFG (1987) Habitat complexity and coral reef fish diversity and abundance on Red Sea fringing reefs. Mar Ecol Prog Ser 41:1-8

Rodriquez SR, Ojeda FP, Inestrosa NC (1993) Settlement of benthic marine invertebrates. Mar Ecol Prog Ser 97:193-207

Rogers CS, III HCF, Gilnack M, Beets J, Hardin J (1984) Scleractinian coral recruitment patterns at Salt River Submarine Canyon, St. Croix, U.S. Virgin Islands. Coral Reefs 3:69-76

Rotjan RD, Lewis SM (2005) Selective predation by parrotfishes on the reef coral *Porities asteroides*. Mar Ecol Prog Ser 305:193-201

Sale PF, Doherty PJ, Eckert GJ, Douglas WA, Ferrell DJ (1984) Large scale spatial and temporal variation in recruitment to fish populations on coral reefs. Oecologia (Berl) 64:191-198

Sammarco PW, Carleton JH (1981) Damselfish territoriality and coral community structure: reduced grazing, coral recruitment, and effects on coral spat. Proc 4th ICRS Manila 2:525-535

Scott PJB, Moser KA, Risk MJ (1988) Bioerosion of concrete and limestone by marine organisms: A 13 year experiment from Jamaica. Mar Poll Bull 19:219-222

Semon K, Paul V, Clementz M, Trapp JM, Arthur K (2008) Evaluation of biotic and abiotic factors that facilitate and maintain cyanobacteria blooms and phase shifts in coral communities. Proc 11th ICRS Ft Lauderdale:69

Sheppard C, Teleki K, Turner J (2000) Measuring change and recovery in reef ecosystems Coral degradation in the Indian Ocean: Status Report 2000 138-140

Sherman RL (2000) Studies on the roles of reef design and site selection in juvenile fish recruitment to small artificial reefs. Ph.D. Dissertation, Nova Southeastern University, p173

Sherman RL, Gilliam DS, Spieler RE (1999) A preliminary examination of depth associated spatial variation in fish assemblages on small artificial reefs. J Appl Ichthyol 15:116-121

Sherman RL, Gilliam DS, Spieler RE (2001) Effects of refuge size and complexity on recruitment and fish assemblage formation on small artificial reefs. Proc 52nd Gulf Carib Fish Instit 52:455-467

Sherman RL, Gilliam DS, Spieler RE (2002) Artificial reef design: void space, complexity, and attractants. ICES J Mar Sci 59:S196-S200

Shima JS (2002) Mechanisms of density- and number-dependent population regulation of a coral-reef fish. Mar Freshwater Res 53:175-179

Shima JS, Osenberg CW, Mary CMS (2008) Quantifying site quality in a heterogeneous landscape: recruitment of a reef fish. Eco 89:86-94

Shulman MJ (1985a) Coral reef fish assemblages: intra- and interspecific competition for shelter sites. Envir Biol Fishes 13:81-92.

Shulman MJ (1985b) Recruitment of coral reef fishes: effects of distribution of predators and shelter. Eco 66:1056-1066

Smith SR (1992) Patterns of coral recruitment and post-settlement mortality on Bermuda's reefs: comparisons to Caribbean and Pacific reefs. Amer Zool 32:663-673

Soloviev AV, Thomson TL, Nemeth L, Campbell CB, Weisberg RH, Luther ME, Cole, R., Donovan J (2001) Data from the NSU/USF Environmental Array (25 June 1999-3 October 2000) including the SFOMC "Four-dimensional and adverse weather experiments 1999 and 2000"

Soong K (1993) Colony size as a species character in massive reef corals. Coral Reefs 12:77-83

Spieler RE, Gilliam DS, Sherman RL (2001) Artificial reefs and coral reef restoration: what do we need to know to know what we need? Bull Mar Sci 69:1013-1030

Sutton M (1983) Relationships between reef fishes and coral reefs. In: Barnes DJ (ed) Perspectives on Coral Reefs. Aust. Inst. Marine Sci., Manuka, pp248-255

Szmant AM (1986) Reproductive ecology of Caribbean reef corals. Coral Reefs 5:43-54

Szmant AM (1991) Sexual reproduction by the Caribbean reef corals *Montastrea annularis* and *M. cavernosa*. Mar Ecol Prog Ser 74:13-25

Thanner SE, McIntosh TL, Blair SM (2006) Development of benthic and fish assemblages on artificial reef materials compared to adjacent natural reef assemblages in Miami-Dade County, Florida. Bull Mar Sci 78:57-70

van Moorsel GWNM (1988) Early maximum growth of stony corals (Scleractinia) after settlement on artificial substrata on a Caribbean reef. Mar Ecol Prog Ser 50:127-135

Vermeij MJA (2005) Substrate composition and adult distribution determine recruitment patterns in a Caribbean brooding coral. Mar Ecol Prog Ser 295:123-133

Wallace CC, Bull GD (1981) Patterns of juvenile coral recruitment on a reef front during a spring-summer spawning period. Proc 4th ICRS Manila 2:345-350

Weinstein MP, Teal JM, Balletto JH, Strait KA (2001) Restoration principles emerging from one of the world's largest tidal marsh restoration projects. Wetlands Eco and Mgmt 9:387-407

Wellington GM (1982) Depth zonation of corals in the Gulf of Panama: control and facilitation by resident reef fishes. Ecol Mono 52:223-241

Wieczorek SK, Todd CD (1989) Inhibition and facilitation by bryozoan and ascidian settlement by natural multi-species biofilms: effects of film age and the roles of active and passive larval attachment. Mar Biol:463-473

Woodley JD, Clark JR (1989) Rehabilitation of degraded coral reefs. Proc 6th Symp on Coastal and Ocean Mgmt:3059-3075

Yap HT (2000) The case for restoration of tropical coastal ecosystems. Ocean and Coast Mgmt 43:841-851

Zar JH (1996) Biostatistical analysis. Prentice Hall, NJ

Zarull MA, Hartig JH (2001) Aquatic ecosystem rehabilitation: targets, actions, responses. Aquat Eco Health and Mgmt Soc 4:115-122